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Proceedings of the Symposium on the Development Use and Value of Water Quality Criteria and Standards at Washington, DC on June 23-25, 1980

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PROCEEDINGS

Symposium on the Development, Use and Value of Water Quality Criteria and Standards at Washington, D. C., June 23-25, 1980

Cosponsored By:
The Maryland Dept. of Natural Resources, annapolis.
The Virginia State Water Control Board, Richmond.
The District of Columbia, Dept. of
Environmental Services Washington.
The George Washington University School Washington, DC.
of Engineering and Applied Science

Edited By:

Bruce A. Bell and Charles R. Bostater

Grant No. T901119010

Project Officer

William E. Fox
Office of Water Planning and Standards
Criteria and Standards Branch
U.S. Environmental Protection Agency
Washington, D. C. 20460

OFFICE OF WATER PLANNING AND STANDARDS
STANDARDS AND CRITERIA BRANCH
U.S. Environmental Protection Agency
Washington, D. C. 20460

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Bruce A. Bell, and Charles R. Bostater.

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A Symposium on The Development, Value and Use of Water Quality Criteria and Standards was held at The George Washington University, in Washington, D.C. on June 23 through 25, 1980. The purpose of the Symposium was to provide a forum for policy and scientific presentations concerning the current and future role of water quality criteria and standards in water quality management. The primary areas covered are: Approaches to Water Quality Management; Implementation of Water Quality Criteria and Standards; Human Health Effects in Development of Criteria and Standards; Protection of Aquatic Biota in Developing Criteria and Standards; Monitoring, Sampling and Analysis to Support Criteria and Standards; Predicting Concentrations of Chemicals in the Environment.

KEYWORDS: *Policies, *Water quality management, *Meetings, *Water quality criteria.

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FORWARD

These proceedings result from the formal papers and panel discussions from the Symposium on the Development, Use and Value of Water Quality Criteria and Standards. In the past, few symposia have focused on the utility and practicality of water qualtiy criteria and standards. Recently emphasis has been placed on source control standards or effluent permit limitations. This change in regulatory emphasis has been reflected in the policies involving water quality standards. Although criteria are meant to be goals based on scientific judgement, the states are expected to adopt standards which are not less adequate than these goals. Thus, a conflict may arise when states attempt to adopt standards based on monitoring, technology, costs, etc.

This symposium has provided a forum where the various aspects of criteria and standards use and application were discussed. Through the use of panel discussions various viewpoints were presented and discussed. Issues of interest were documented and controversial issues discussed in an open atmosphere. Through the efforts of those participating, attention was focused on the development and utilization of practical, useable water quality standards and criteria.

PREFACE

These proceedings are intended to disseminate the results of the interactions between engineers, scientists and managers involved with research, evaluation, development, administration and utilization of water quality standards and criteria. Participants were from the United States and abroad. This symposium, sponsored by the U.S. Environmental Protection Agency, The George Washington University, The State of Maryland, The District of Columbia and the Commonwealth of Virginia, contained presentations on the State of the Art of research related to water quality standards and criteria, and panel discussions on the use and value of standards and criteria.

The papers in these proceedings are arranged as they were presented at the symposium. The prepared statements of the panelists are included along with the discussions during the panels where available. All of the presentations are related to water quality standards and criteria.

The papers are printed here basically as received from the authors. They do not necessarily reflect the policies and opinions of the U.S. Environmental Protection Agency or the co-sponsors of the symposium. These proceedings will hopefully be of use to the engineering, scientific and administrative community concerned with water quality standards and criteria.

DISCLAIMER

These proceedings have been reviewed by the U. S. Environmental Protection Agency and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U. S. Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

ABSTRACT

A Symposium on The Development, Value and Use of Water Quality Criteria and Standards was held at The George Washington University, in Washington, D. C. on June 23 through 25, 1980. The purpose of the Symposium was to provide a forum for policy and scientific presentations concerning the current and future role of water quality criteria and standards in water quality management and to provide an opportunity for engineers, managers and scientists to discuss the relation between criteria and legal standards and the role of sampling, analysis and modeling in setting water quality objectives. These proceedings are a compilation of the papers presented by panelists and audience participation in panels. The primary areas covered are:

(1) Approaches to Water Quality Management.
(2) Implementation of Water Quality Criteria and Standards (panel).
(3) Human Health Effects in Development of Criteria and Standards.
(4) Protection of Aquatic Biota in Developing Criteria and Standards.
(5) Monitoring, Sampling and Analysis to Support Criteria and Standards.
(6) Predicting Concentrations of Chemicals in the Environment.
(7) Future of Criteria and Standards (panel).

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APPROACHES TO WATER QUALITY MANAGEMENT

SECTION I

WATER QUALITY OBJECTIVES ACTIVITIES IN THE DEPARTMENT OF THE ENVIRONMENT

Adrian Demayo and Margaret C. Taylor

Water Quality Branch
Environment Canada
Ottawa, Canada K1A 0E7

ABSTRACT

In the first part, the paper describes the organization of the Department of the Environment of the canadian federal government and the role and the activities of its Water Quality Branch (WQB).

In the second part the paper describes the activities in the water quality objectives field carried out in the WQB and the approach used until now to establish water quality objectives for some rivers and lakes in Canada.

1. INTRODUCTION

The federal and provincial governments' activities in relation to water in Canada is derived fundamentally from the British North America Act of 1867 (Canada's Constitution). The act provides for provincial jurisdiction relating to proprietary rights, which includes the use, regulation and development of water resources within their boundaries. The Act gives the power to the federal government to legislate over a series of specific matters related to water in areas where there is a "significant national interest" such as boundary and international waters, interprovincial waters and waters in national parks, territories or other federal lands. Issues of national interest in which the federal government has specific powers and which are related to water are: fisheries, navigation, migratory birds, agriculture and regional development. Examples of major legislation approved by the Parliament of Canada which are related to water are: the Boundary Waters Treaty Act (1909), the International Rivers Improvements Act (1955), the Canada Water Act (1970), the Arctic Waters Pollution Prevention Act (1970), the Department of the Environment Act (1970), and the Fisheries Act (as amended in 1977).

2. ORGANIZATION AND ROLE OF THE DEPARTMENT OF THE ENVIRONMENT

The Department of the Environment (DOE), created by an act of Parliament in 1970, has undergone several reorganizations. Its present structure is shown in figure 1. Table 1 shows the budgetary allocations for the current fiscal year, i.e. April 1, 1980 to March 31, 1981.

The objectives of the Environmental Services Program of the Department of the Environment are (Government of Canada "Estimates for the fiscal year ending March 31, 1981"):

> "To promote and undertake programs to protect and enhance the quality of the environment, and programs designed to improve the management and sustained economic utilization of the forest, wildlife and inland water resources of the nation".

In the area of water, besides the general responsibility for planning and managing water resources of significant national interest and the power to legislate in a number of matters that affect water, the federal government also has a mandate for "... collecting data and maintaining inventories relating to surface and subsurface water, pollutants and contaminants, fisheries and aquatic ecosystems, atmospheric water, and snow and ice" (Department of the Environment - Federal Policy Statement on Inland Waters 1978):

FIGURE 1. THE ORGANIZATION OF THE DEPARTMENT OF THE ENVIRONMENT

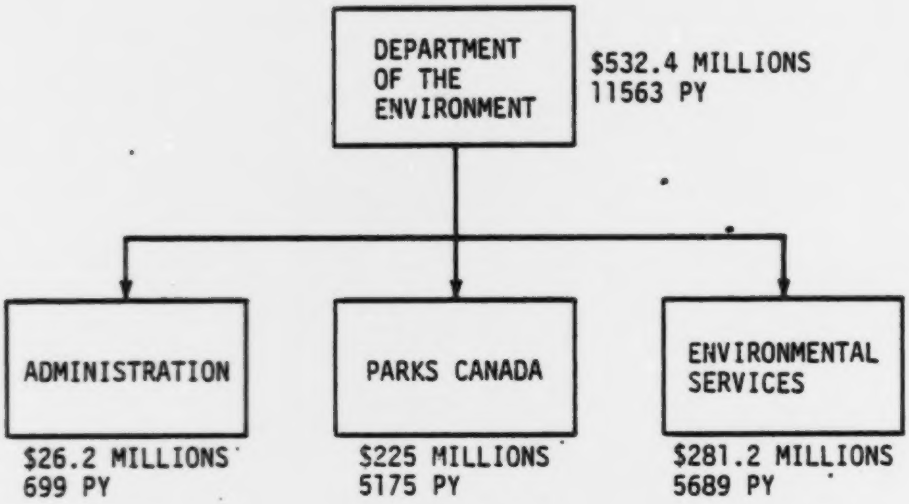


TABLE 1. BUDGETARY ESTIMATES
IN THE
DEPARTMENT OF THE ENVIRONMENT
FISCAL YEAR 1980 - 1981

PROGRAM	ESTIMATED EXPENDITURES ($ MILLIONS)	TOTAL PERSON-YEARS AUTHORIZED
ADMINISTRATION	26.2	699
ENVIRONMENTAL SERVICES		
ENVIRONMENTAL CONSERVATION (MANAGEMENT)	64.2	1580
FORESTRY	39.6	1029
ATMOSPHERIC ENVIRONMENT	103.7	2333
ENVIRONMENTAL PROTECTION	32.9	747
OTHER EXPENDITURES (E.G. EMPLOYEE BENEFIT PLANS, PAYMENTS TO PROVINCES)	40.8	
PARKS CANADA	225.0	5175
TOTAL	532.4	11563

REF. GOVERNMENT OF CANADA. ESTIMATES FOR THE FISCAL YEAR ENDING MARCH 31, 1981.

In carrying out its mandate the Department of the Environment is addressing most of the problems through a joint federal-provincial approach. This policy recognizes that the federal government and the provincial governments each have distinct but interdependent responsibilities for water in Canada. The federal government's concern with respect to water is to ensure, in cooperation with the provinces, that the water resources of Canada are protected and used for the greatest social and economic benefit of all Canadians, while striking a balance between the needs of present and future generations.

The water quality objectives activity, which is the subject of this paper, is located in the Environmental Conservation Program. Some of the other activities carried out within this program are:

- hydrometric data collection
- water quality data collection
- research in water related areas such as: pathways of contaminants through the ecosystem, analytical methodology, limnology
- migratory birds conservation and protection

The organizational structures of the Environmental Services Program and of the Environmental Conservation Service are shown in figures 2 and 3. Figure 4 shows, schematically, the organization of the Inland Waters Directorate, and finally, figure 5 shows the organization of the Water Quality Branch (WQB). The Water Quality Objectives group is part of this branch. The WQB across Canada has approximately 130 PY of which 24 PY are located in the National Capital Region (NRC). The Water Quality Objectives Group has a complement of 6 permanent positions, (5 professionals and one secretary) plus one temporary position (technical assistant).

3. ROLE OF THE WATER QUALITY BRANCH

Within the broad statement of the role of the federal government, the specific objective of the Water Quality Branch is "to provide scientific and technical information on water quality to managers of water quality and pollution control programs, and to the public at large." As a part of this role the Branch provides ambient water quality data, interpretative information, and scientific reports in support of:

(a) federal water and renewable natural resources management and control programs;

(b) environmental assessment and review procedures;

FIGURE 2. THE ORGANIZATION OF THE ENVIRONMENTAL SERVICES PROGRAM

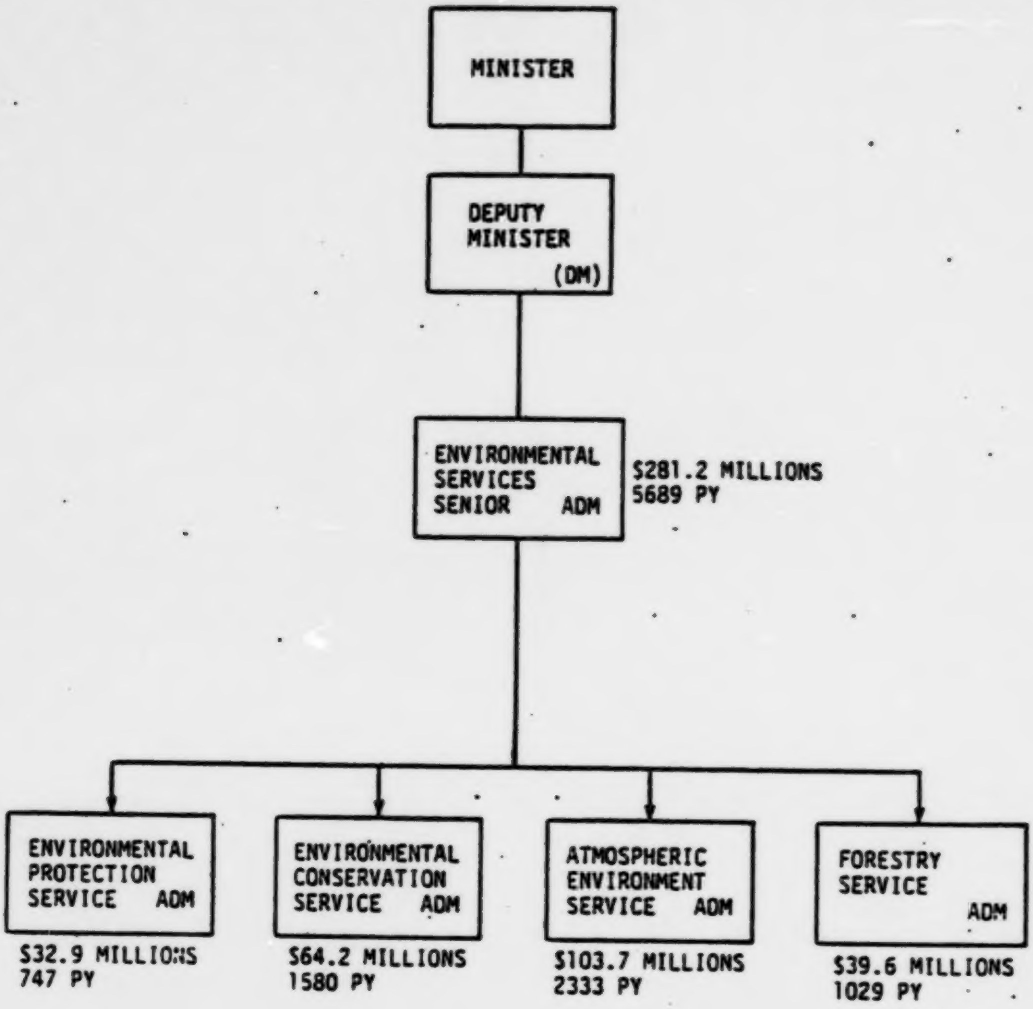


FIGURE 3. THE ORGANIZATION OF THE ENVIRONMENTAL CONSERVATION SERVICE (ECS)

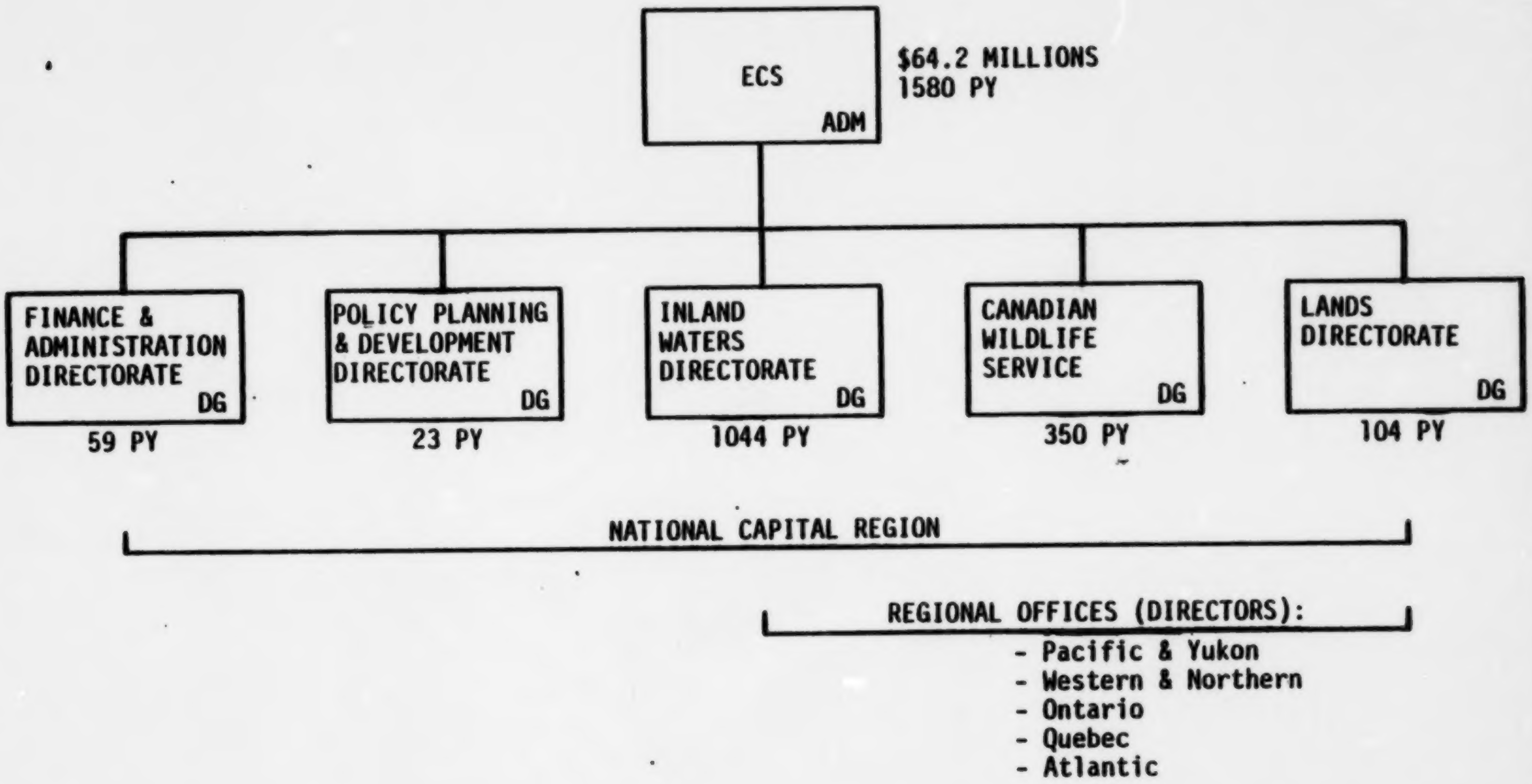


FIGURE 4. THE ORGANIZATION OF INLAND WATERS DIRECTORATE (IWD)

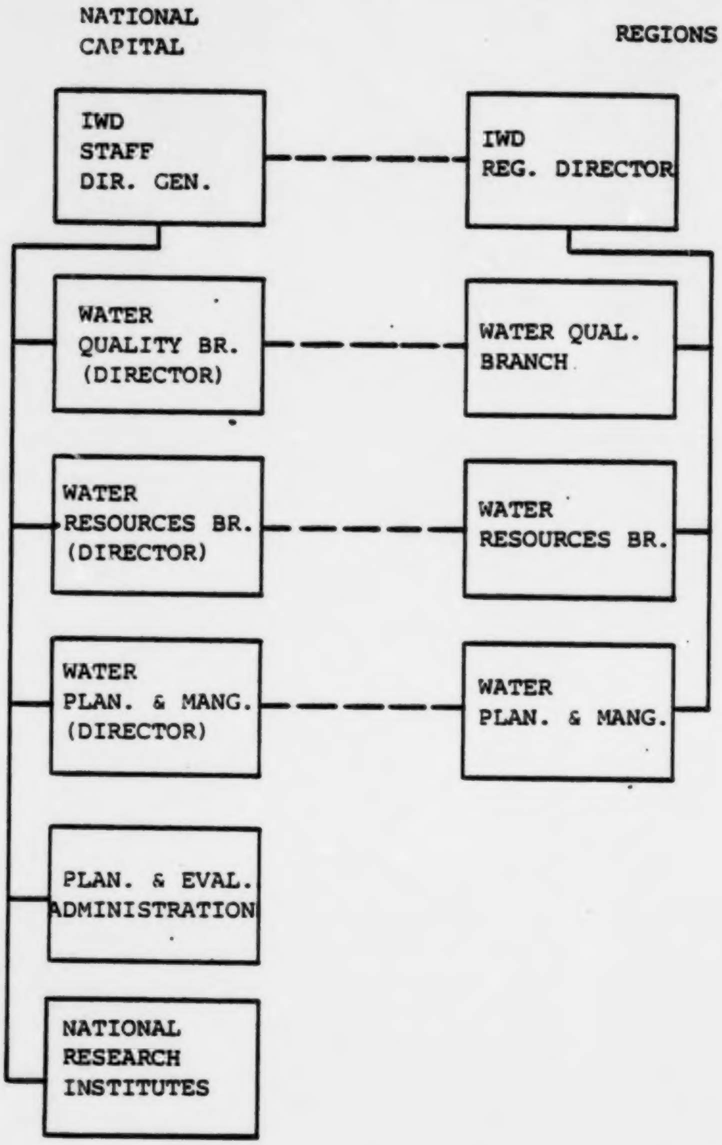
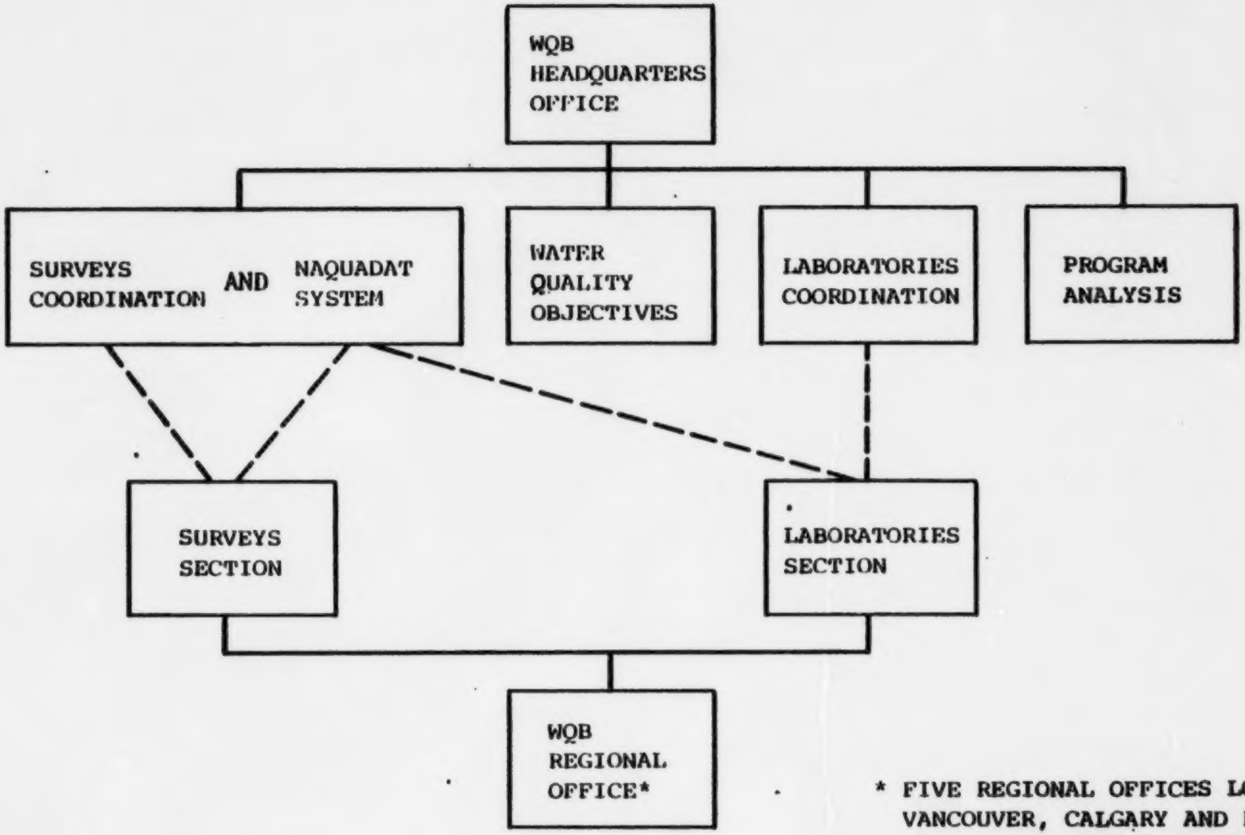

NATIONAL CAPITAL
REGIONS
IWD STAFF DIR. GEN.
IWD REG. DIRECTOR
WATER QUALITY BR. (DIRECTOR)
WATER QUAL. BRANCH
WATER RESOURCES BR. (DIRECTOR)
WATER RESOURCES BR.
WATER PLAN. & MANG. (DIRECTOR)
WATER PLAN. & MANG.
PLAN. & EVAL. ADMINISTRATION
NATIONAL RESEARCH INSTITUTES

FIGURE 5. SCHEMATIC ORGANIZATION OF THE HEADQUARTERS AND THE REGIONAL OFFICES OF THE WATER QUALITY BRANCH (WQB)



* FIVE REGIONAL OFFICES LOCATED IN VANCOUVER, CALGARY AND REGINA, BURLINGTON, MONTREAL, AND MONCTON

(c) legislation and research activities; and

(d) federal-provincial and international agreements and commitments.

Activities of the Water Quality Branch

The objective stated above is accomplished by carrying out the following activities in the Regions and at Headquarters:

Regional Offices:

In the regional offices, the following activities are carried out:

(a) Design of water quality surveys, which includes the determination of the location of the sampling points, the frequency of sampling and the characteristics to be measured.

(b) Collection of water, sediment, and sometimes biota samples from rivers, lakes, estuaries, etc.

(c) Analysis of these samples for a series of physical and chemical characteristics.

(d) Reporting water quality data and writing interpretative reports.

Headquarters:

At Headquarters, the following activities are conducted:

(a) Development of water quality criteria and objectives.

(b) Operation and maintenance of the NAQUADAT (National Water Quality Data Bank).

(c) Support of the regional offices in areas such as writing interpretative reports, design of surveys, quality control and efficiency evaluation of laboratories.

(d) Advice to senior management (Director General, Assistant Deputy Minister) in water quality matters.

4. THE DEVELOPMENT OF WATER QUALITY OBJECTIVES IN CANADA

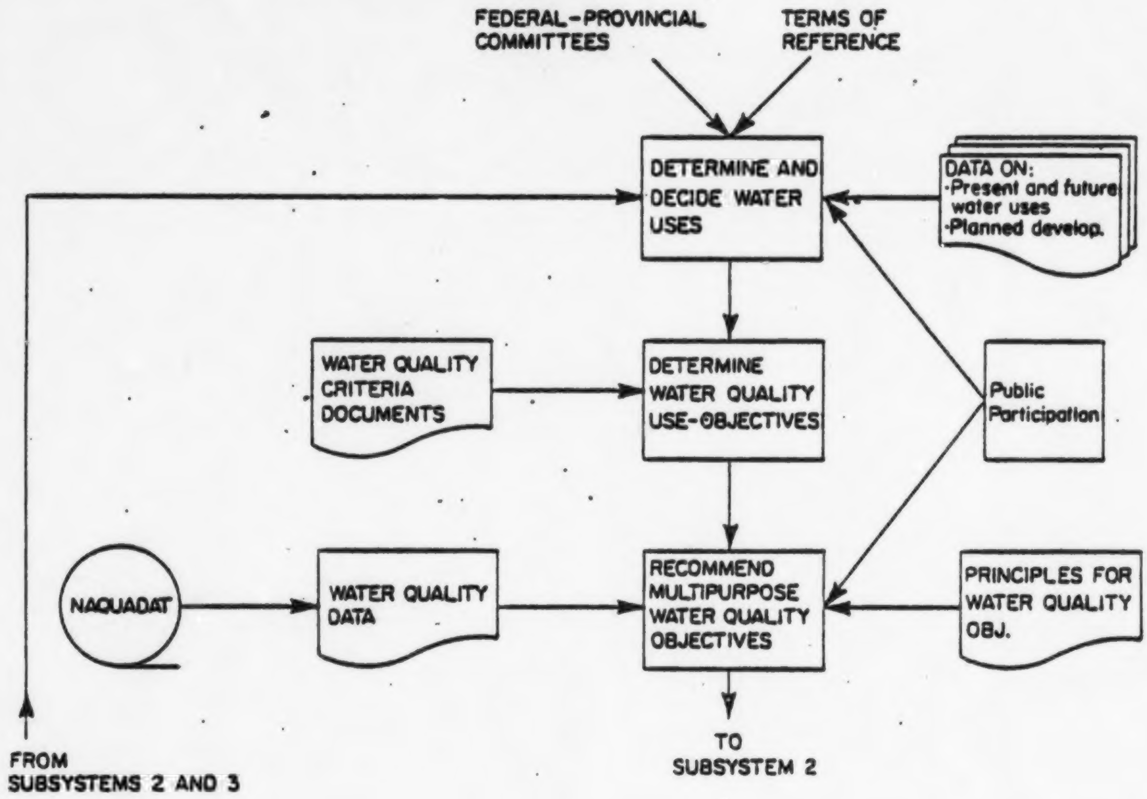
As mentioned before, the Canadian federal government is following a policy of a joint federal-provincial approach in the management of water resources in which there is a federal interest. This paper will not deal with the water quality objectives in waters which are strictly of provincial interest.

FIGURE 6

SCHEMATIC REPRESENTATION OF A SYSTEM FOR ESTABLISHING WATER QUALITY OBJECTIVES FOR A SPECIFIC WATER BODY

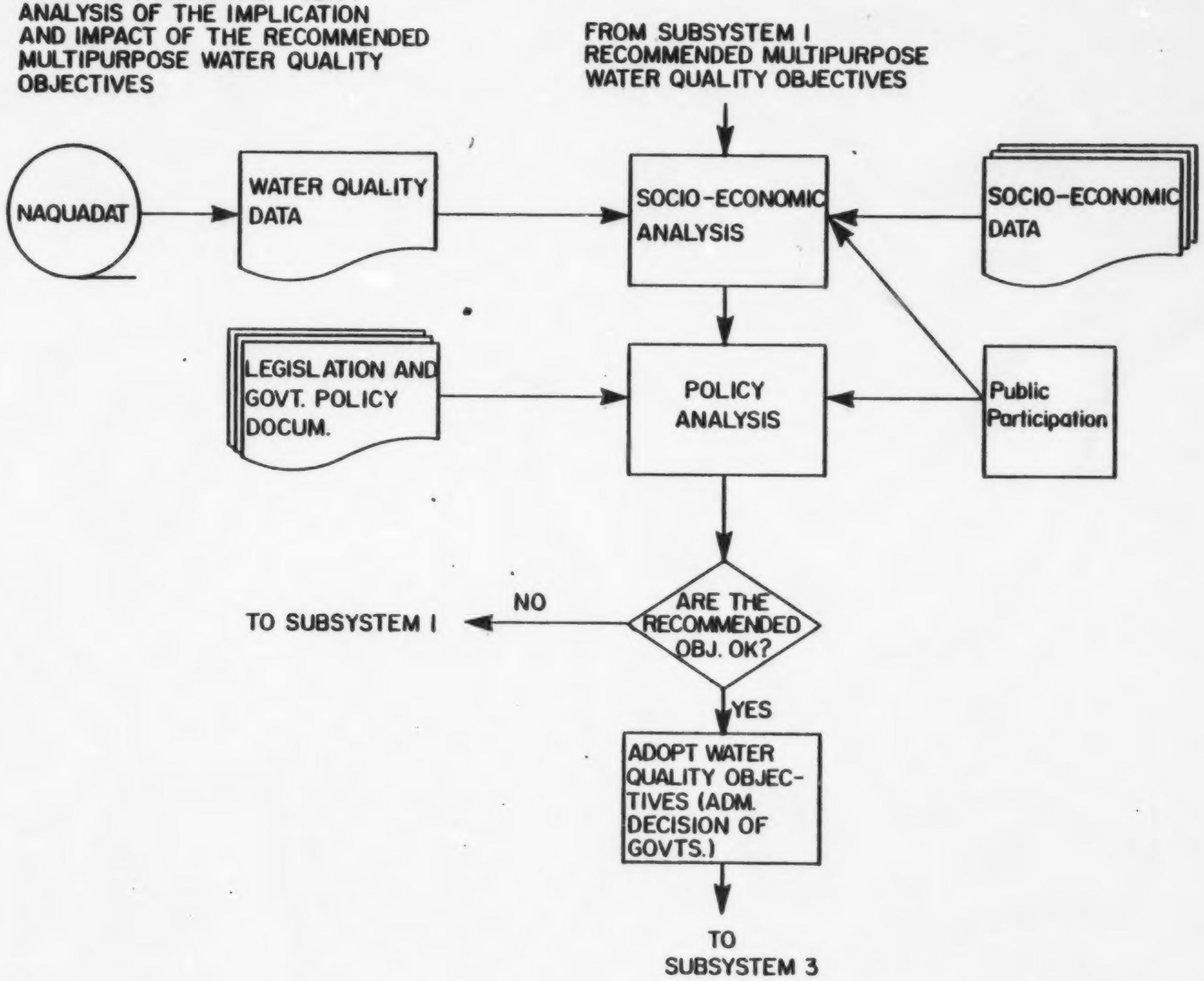
SUBSYSTEM I

THE DEVELOPMENT OF A RECOMMENDED
SET OF MULTIPURPOSE WATER QUALITY OBJECTIVES


FEDERAL-PROVINCIAL COMMITTEES
TERMS OF REFERENCE
DETERMINE AND DECIDE WATER USES
DATA ON: ·Present and future water uses ·Planned develop.
WATER QUALITY CRITERIA DOCUMENTS
DETERMINE WATER QUALITY USE-OBJECTIVES
Public Participation
NAQUADAT
WATER QUALITY DATA
RECOMMEND MULTIPURPOSE WATER QUALITY OBJECTIVES
PRINCIPLES FOR WATER QUALITY OBJ.
FROM SUBSYSTEMS 2 AND 3
TO SUBSYSTEM 2

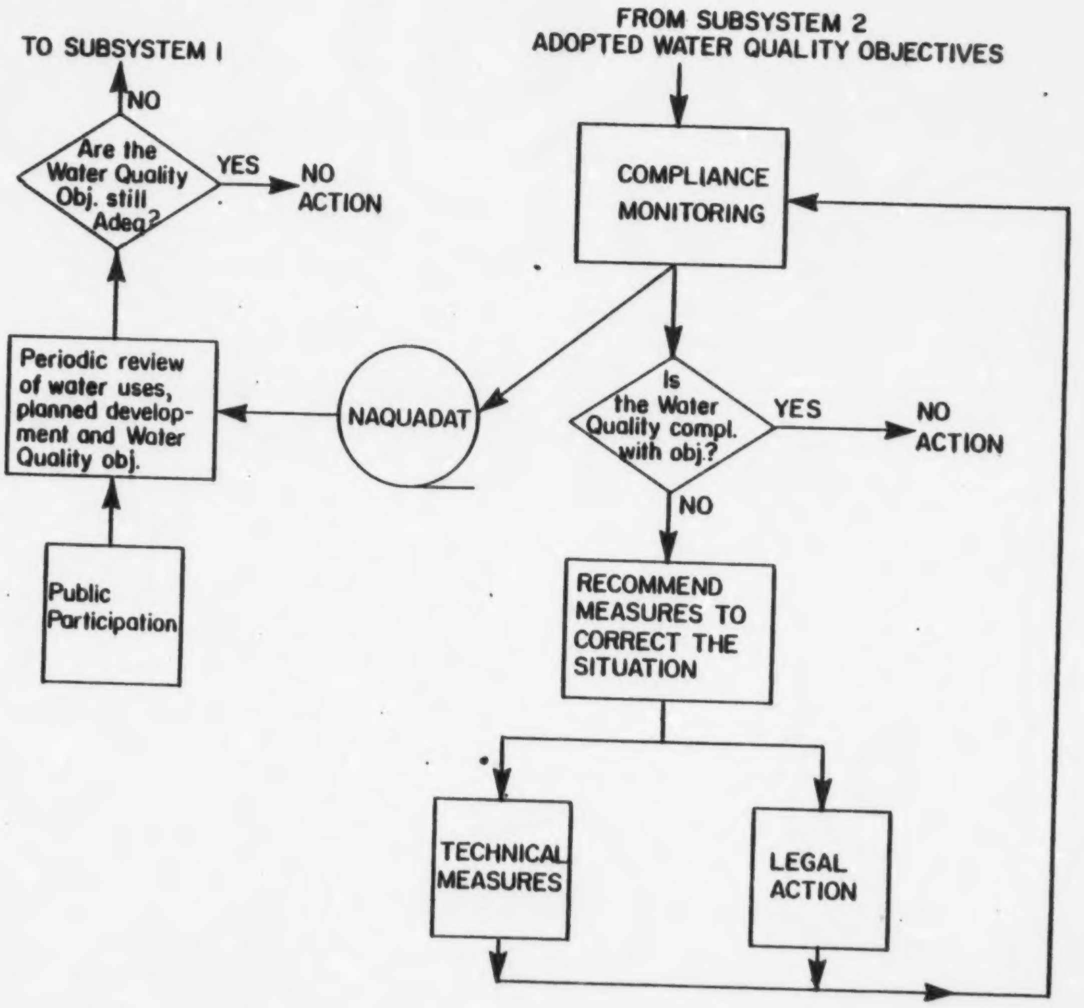
SUBSYSTEM 2

SOCIO-ECONOMIC AND POLICY ANALYSIS OF THE IMPLICATION AND IMPACT OF THE RECOMMENDED MULTIPURPOSE WATER QUALITY OBJECTIVES



SUBSYSTEM 3

COMPLIANCE MONITORING AND PERFORMANCE REVIEW


TO SUBSYSTEM I
NO
Are the Water Quality Obj. still Adeq?
YES
NO ACTION
Periodic review of water uses, planned development and Water Quality obj.
Public Participation
NAQUADAT
FROM SUBSYSTEM 2
ADOPTED WATER QUALITY OBJECTIVES
COMPLIANCE MONITORING
Is the Water Quality compl. with obj.?
YES
NO ACTION
NO
RECOMMEND MEASURES TO CORRECT THE SITUATION
TECHNICAL MEASURES
LEGAL ACTION

The following definitions are used in the Water Quality Branch:

- criteria: scientific information (e.g. concentration-effect data) which is used to recommend water quality objectives.
- objective: a designated concentration of a constituent that, when not exceeded, will protect a specific water use (e.g. drinking, aquatic community), in a specific body of water with an adequate degree of safety. The objective is based on scientific criteria, local natural conditions, and social and environmental considerations.
- standard: the concentration of a constituent or a narrative statement established under statutory authority, e.g., legally enforceable.

Figure 6 shows the process which has been followed in Canada in establishing water quality objectives in a specific body of water.

The process of developing water quality objectives is triggered by one or a combination of several factors such as: (i) political decisions, which either follow a predetermined government policy line, or which are taken in response to a specific case or demand; (ii) technical or scientific reasons such as the management of the water or fisheries resource; (iii) in response to public opinion. In a typical case, at least until now, a comprehensive study in a given basin is undertaken, which includes, amongst other things, a water quality component. Usually, part of the water quality component is the development of water quality objectives. As shown, several elements enter in the development of water quality objectives. They are discussed next.

4.1 Scientific Criteria.

Scientific criteria relate the concentration of a substance to its effects on various forms of life, e.g. terrestrial (human, livestock, plants) and aquatic (plants, fish, invertebrates). These effects can be either beneficial or toxic. The types of adverse effects that must be considered are, for example, on reproduction, growth, resistance to disease, the possibility of carcinogenesis, blood disorders, histological and physiological changes and the accumulation in body tissues. The scientific criteria also provide, when available, information on topics such as the pathway(s) by which substances enter the environment, transfer rates between various compartments of the ecosystem, and fate in the environment. The scientific information is usually summarized in the so-called criteria documents. The Water Quality Branch (WQB), as well as other groups in Canada and other countries, is producing such documents. Fig. 7, presents, in a flow-diagram, the process used to write the criteria documents in the WQB. The WQB documents are published under the title of "Guidelines for Surface Water Quality" and they include so far a preamble, arsenic, cadmium, chromium, mercury, nickel, selenium and silver. Documents for lead, copper and zinc will be published during the current year.

The "Guidelines" discuss: sources, occurrence, natural distribution, pathways for entering the environment, chemical forms, production, consumption, uses and, finally, the effects on humans, laboratory animals, aquatic life, wildlife, livestock and plants. The water uses for which objectives are recommended in the "Guidelines" are: aquatic life, public raw water supply, irrigation, livestock watering, the food processing industry, and recreation and aesthetics.

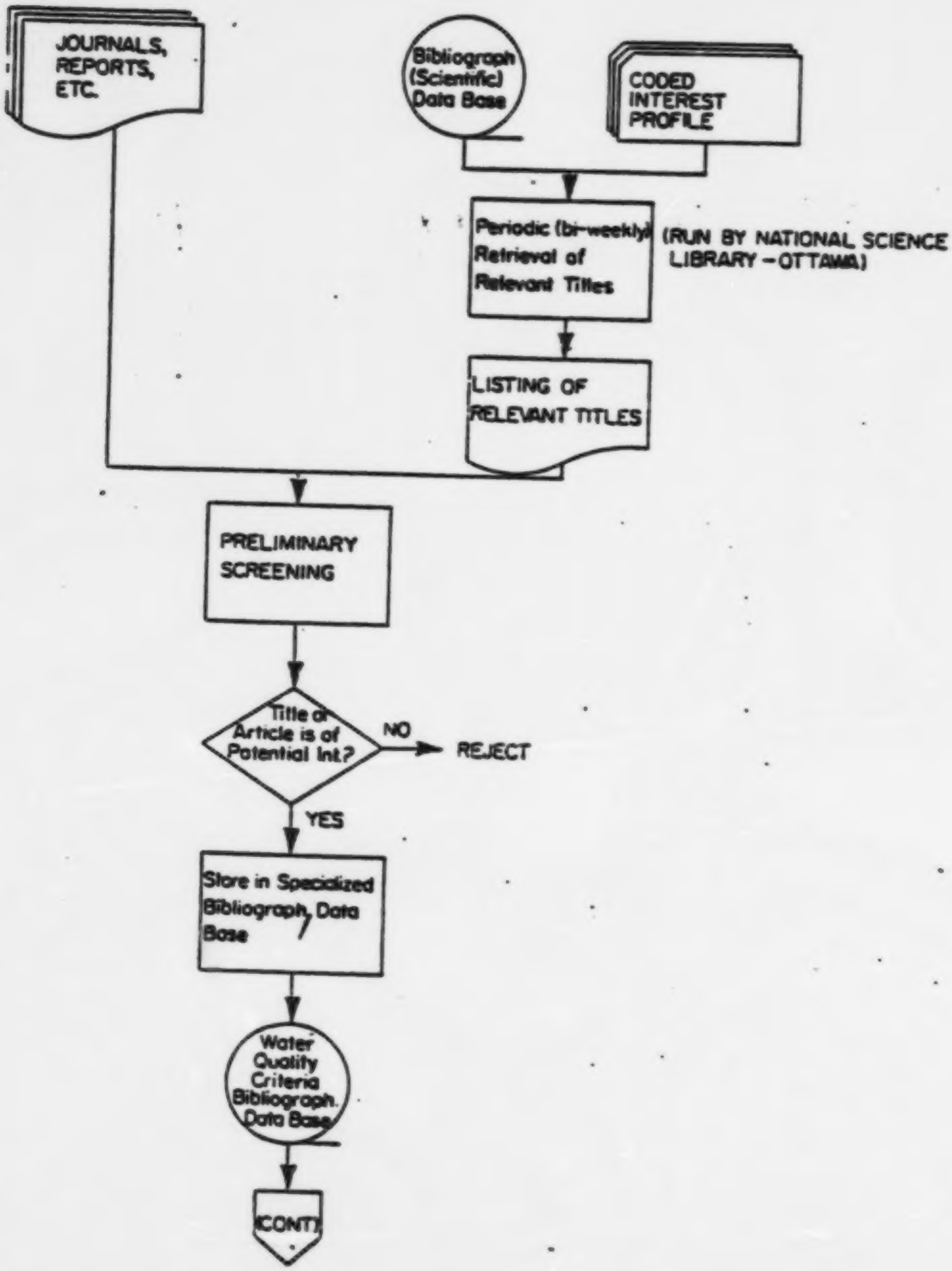
In writing the "Guidelines" we try to use the "best available information". In using the results of research there are certain basic questions which have to be asked about the data, such as the suitability of the method to answer the hypothesis of the researcher, whether or not all factors were considered, e.g., the quality of the experimental water, viability of the test organisms; whether or not the planning of the experiments and their statistical analysis were sound, and whether or not the experiments were relevant to the Canadian environment.

Although the number of papers related to the fate of contaminants and their effects on the environment published each year is very large, the gap they have to fill is much larger. Some of the points which are particularly important to the determination of the water quality objectives are, for example:

- the distribution of a substance between various compartments of the ecosystem and the transfer rates between them, (e.g. sediment-water-air).
- chemical and physical forms of a substance in the environment and their biological availability.
- synergism and antagonism between various substances.
- stress and acclimation of organisms.
- sublethal effects of substances.

FIGURE 7

SCHEMATIC REPRESENTATION OF THE SYSTEM FOR DEVELOPING WATER QUALITY CRITERIA DOCUMENTS


JOURNALS, REPORTS, ETC.
Bibliograph (Scientific) Data Base
CODED INTEREST PROFILE
Periodic (bi-weekly) Retrieval of Relevant Titles
(RUN BY NATIONAL SCIENCE LIBRARY - OTTAWA)
LISTING OF RELEVANT TITLES
PRELIMINARY SCREENING
Title or Article is of Potential Int.?
NO
REJECT
YES
Store in Specialized Bibliograph. Data Base
Water Quality Criteria Bibliograph. Data Base
(CONT)

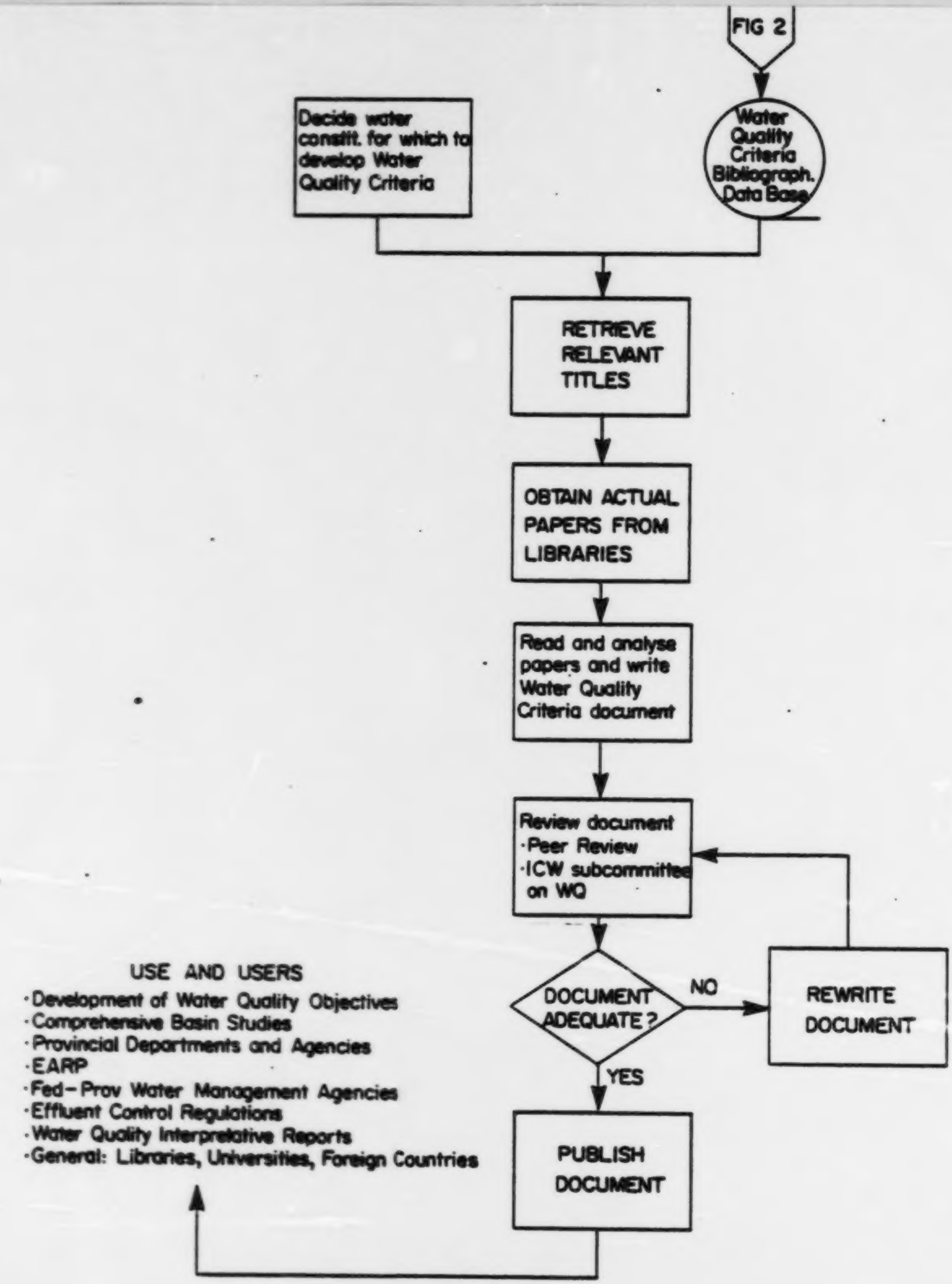


FIG 2
Decide water constit. for which to develop Water Quality Criteria
Water Quality Criteria Bibliograph. Data Base
RETRIEVE RELEVANT TITLES
OBTAIN ACTUAL PAPERS FROM LIBRARIES
Read and analyse papers and write Water Quality Criteria document
Review document
·Peer Review
·ICW subcommittee on WQ
DOCUMENT ADEQUATE ?
NO
REWRITE DOCUMENT
YES
PUBLISH DOCUMENT
USE AND USERS
·Development of Water Quality Objectives
·Comprehensive Basin Studies
·Provincial Departments and Agencies
·EARP
·Fed–Prov Water Management Agencies
·Effluent Control Regulations
·Water Quality Interpretative Reports
·General: Libraries, Universities, Foreign Countries

Recognizing the fact that it will be very long before all such information becomes available, certain procedures are being used in the WQB to fill particular gaps. For example, application factors (AF) have been used to derive what is, hopefully, safe levels from acute, short-term studies for aquatic life. An AF of 0.01 is used if the pollutants are known to persist or to biomagnify through the food web. A factor of 0.05 is used with non-persistent or non-bioaccumulative pollutants when only acute data are available. When sublethal studies are very sparse but a small amount of information is available, an AF of 0.2 is applied to the 96-h LC_{50} results. In the case of silver sufficient information was available to enable a comparison of a "safe level" determined by this method with that derived from some sublethal, long-term studies. The "safe levels" derived from both methods were very close and this gave a measure of confidence in the recommended water quality objectives.

Another example of an interim decision until more information becomes available, is that of recommending water quality objectives on "total" quantities rather than "dissolved", extractable" or some other operational definition of the form of a substance in water . It was felt that until such time when we know more about the biological availability of various chemical forms and the physical and chemical processes which affect these forms, and until routine analytical methodology to measure these various forms becomes available, the recommended objectives should be based on "total" concentrations, thus in fact incorporating an additional safety factor into the objective.

4.2 Socio-Economic Factors.

These factors include legislation, federal-provincial relations, international obligations, the economic situation and economic policy.

Another very important element, which could be considered, either separately or under this heading, is the "effluent limitations". There is agreement amongst water quality-water pollution specialists that effluent limitations are related to water quality objectives. However, there is no agreement and not too much practical experience on exactly how one can relate the two sets of values. The common idea is to establish water quality objectives and then, by some calculations, obtain the effluent concentrations which will result in values below or equal to the recommended objectives in the receiving waters. Sufficient consideration has not been given, yet, to the opposite question of the role which effluent limitations should play in the development of water quality objectives as part of the economic factors, for example.

4.3 Background (Natural, Predevelopment) and Existing Water Quality).

Although there is agreement that the existing water quality should be considered when establishing water quality objectives, there is no agreement regarding the values one should use to define the water quality, e.g., maximums, means, medians or some other percentile. The sampling technique, frequency of sampling; parameters, physical form (e.g. total, dissolved) and chemical species to be measured; and field vs. laboratory measurements are some of the other issues related to the determination of the existing water quality which must be considered in order to obtain facts which can be used in the setting and then monitoring of water quality objectives.

Regarding the "background" water quality there is no agreement on how the corresponding values can be obtained and used in the setting of water quality objectives, especially when reliable past data are not available. This being the case, it is questionable whether or not the expense and the manpower effort involved in obtaining "natural" water quality is justified by its contribution to the final water quality objectives. A workshop on the subject of water quality objectives held in the Water Quality Branch in May 1980, concluded that the idea of determining the "predevelopment" water quality in view of using it in recommending water quality objectives should be dropped. The issue of "predevelopment" water quality is important to Canada in the case of rivers crossing provincial boundaries where some agreement must be reached in apportioning the "assimilative capacity" of these rivers.

4.4 Public Participation

Another element in the development of water quality objectives is the participation of the public and interest groups in this process. The usual procedure is to ask for this input through public hearings in one of the final steps before such objectives are approved. This could be an efficient method, but it is questionable, however, whether or not it allows for an effective input from such groups. It might be that for an effective input, these groups should be involved in the process at an earlier step. Whether or not this is desirable or even possible is a matter for debate.

4.5 The Purpose of Water Quality Objectives

In Canada, water quality objectives are viewed as goals and indicators in water resource management and pollution control. They are of critical importance to the economic and social development of upstream and downstream portions of a basin and because in many cases basins cross interprovincial and international boundaries, both federal and provincial governments view this matter as one of vital concern to them. The establishment of water quality objectives should prevent water quality problems arising in a given basin, lead to a more orderly development and eliminate some of the uncertainties facing developers and government agencies, and thus do away to a large extent with the need for lengthy and costly studies.

5. CONCLUDING REMARKS

Although the Water Quality Objectives Group in the Water Quality Branch has been in existence for approximately 6 years, there is evidence that the process currently used for recommending water quality objectives contains a certain degree of empiricism and ad-hoc methodology. In view of the complexity of the issue and the paucity of relevant information it is very likely that this situation will continue to exist well into the future. One point on which the persons present at the Water Quality Objectives - May workshop agreed, was the fact that each body of water represents a particular and unique case. Therefore, the recommendation of water quality objectives for a particular body of water must follow a careful study of that particular system. Consequently, in order to facilitate this case by case process, it might be that what we should aim for is the accumulation of information related to water quality objectives, the development of some general principles and, eventually, the development of a methodology for arriving at specific water quality objectives. We should not try to develop water quality objectives which will be applied in all parts of the country. We should also recognize that water quality objectives development is only one of the tools the water manager needs in order to manage the water resource in order to obtain the greatest benefit from it.

Finally, throughout this paper water quality objectives were the only ones mentioned. In the last few years we have been moving towards the so-called "ecosystem approach". In this context, a more restricted approach, but perhaps more valid in our present state of knowledge, would be to consider the "aquatic system" which would include more than the water medium, but less than the complete ecosystem. If the "aquatic system" becomes the working unit, the implication is that instead of only water quality objectives, aquatic system objectives be considered. Such objectives would include, when appropriate, recommended concentrations for sediments and aquatic biota. Whether or not this is desirable and possible is also a matter of debate.

AN OVERVIEW OF THE THAMES WATER AUTHORITY'S APPROACH TO THE SUBJECT OF WATER QUALITY AND ITS APPLICATION TO THE SETTING OF QUALITY STANDARDS

Dr. Peter Casapieri

Thames Water Authority
177 Rosebery Avenue
London EC1 R 4TP

SUMMARY

A brief introduction to the historical background relating to pollution control in the United Kingdom has been described in the paper. This was followed by an outline of recent developments which involved total water integration and constituted a major change in organization of the water industry in 1974. Some 1400 water undertakings were co-ordinated into 10 regional water authorities for England and Wales. In addition to this, the United Kingdom became a member of the European Economic Community and hence incurred the treaty obligations inherent in that body regarding satisfying water criteria for a variety of purposes including water supply, wastewater treatment, fisheries and general amenities.

The U.K. use of environmental quality objectives was outlined with the major emphasis placed upon the freshwaters rather than estuarial and coastal. This was largely because, in general, there is a greater knowledge of the objectives and water quality criteria on the freshwater side. A description was given of the state of the rivers in the Thames Regional Water Authority area using a U.K. national river classification. This latter is shown in the main paper as a series of categories related to water use and the quality standards associated. A similar but less detailed assessment of quality·objectives for tidal waters is also given.

The paper discusses how the quality objectives may be achieved on the basis of setting discharge consent conditions for all discharges whether they are water authority or industrial discharges.

Specific mention has been made of the environmental quality objectives for the Thames Estuary and how discharge consents are set and applied.

Finally, the paper poses a question on the missing gaps in our knowledge for setting adequate and scientifically, substantiated water quality criteria associated with desirable environmental quality objectives. The need is to promote further research, on an international basis, and in particular for tidal waters.

HISTORICAL BACKGROUND

The United Kingdom has a long history of controlling water pollution since the evident effects of the rapid urbanization of the country after the Industrial Revolution of the eighteenth and nineteenth centuries. Many of the major rivers, such as the Thames, Mersey, Tyne and Tees, became polluted and various Acts of Parliament were promulgated in an attempt to prevent continuing pollution. Some of the earlier legislation included the Thames Conservancy Act of 1857 which gave authority for the management, improvement and regulation of the river Thames from Staines (only a few kilometres to the west of London) to the Yantlet line which is an arbitrary seaward limit in the outer Thames Estuary. Later Acts extended still further the powers to the whole freshwater Thames catchment whilst pollution control, management and navigation, were ceded to the Port of London Authority in 1908. Subsequently, the Port of London Authority, in the 1960's, was given additional powers to control and limit polluting discharges by the application of flow limitations and restrictions on chemical and toxic waste discharges to the Thames estuary. Thus, the power was available for setting the consent conditions to a discharge, and indirectly quality standards for the recipient waters, but sadly what was diffuse was the firm definition of any quality objectives by which to set the scene.

In the preceding two or three decades, legislation for pollution prevention had been directed largely towards the control of discharges into freshwaters, and only more recently towards control of all discharges into tital and coastal waters generally.

Certain estuaries in the U.K., the Thames included, were fortunate to have pollution control powers; however, this was not of general application to estuaries in the U.K. prior to 1974. Indeed in England and Wales the management of all aspects of the water cycle, including water supply, pollution, prevention, resource management, land drainage and navigation, were looked after by about 1400 separate undertakings, including municipalities, water companies, river authorities, sea fisheries committees, and many ad hoc bodies. This, clearly, was a situation that creaked along and did not allow for any serious integrated water management. Certainly not on a national scale.

RECENT DEVELOPMENTS

The Water Act of 1973 welded most of these undertakings into ten regional water authorities (see figure 1), and Scotland had a similar reorganization which comprised three island councils and seven river purification boards. In the case of England and Wales, since April 1, 1974, the Regional Water Authorities (RWAs) have been responsible to the Secretary of State for the Environment for sewerage, sewage treatment, pollution control, water resources and supply. The RWAs also have responsibilities to the

Minister of Agriculture, Fisheries and Food for land drainage and fisheries (figure 2).

In addition, the Water Act (1973) also established a National Water Council to advise the Government on national policies and to assist the Water Authorities in the efficient performance of their duties. At the same time the Water Research Centre (WRC) was set up funded largely by central government and the Water Authorities to provide technical advice and research services.

In addition to the Water Act which mainly set up the RWAs, the Government has published a Control of Pollution Act (1974) which will enable full control of discharges to all waters and planning of overall strategy. This Act is not yet fully implemented and will over-ride much of the earlier legislation governing pollution prevention and control. The reason for the incomplete implementation of the Act is that in the present economic restraint, central government has to balance environmental advantages against the cost of implememtation, both in the private sector and the public, and the cost might be quite substantial. Thus, the phasing in progressively of sections of the Control of Pollution Act is based on pragmatism in the real world, and not simply the imposition of an ideal which cannot be met.

As if these traumas were not enough, the U.K. entered the European Economic Community in 1973 and, as such, the U.K. government must honour the legal obligations incurred as well as other international ones. Of concern is the Paris Convention, which seeks to eliminate pollution from land-based sources and a number of E.E.C. Directives concerned with:

(a) the quality of bathing water,
(b) pollution caused by certain dangerous substances discharged to the aquatic environment,
(c) waste from the titanium dioxide industry,
(d) quality requirements for waters favourable to shellfish growth,
(e) surface water quality intended for abstraction for drinking water purposes,
(f) quality of water for human consumption,
(g) freshwater fish.

DEFINITION OF ENVIRONMENTAL QUALITY OBJECTIVES

In the U.K., experience of the use of environmental quality objectives has been derived mainly from the application of the control of discharges into freshwaters. The first significant break-through came shortly after the reorganization of the water authorities, and the National Water Council, through the directors of Scientific Services, set up a working party to report on consent conditions for effluent discharges to freshwater streams.[1] The fundamental change was that instead of only commenting on the state of the nation's rivers in a detached manner as the former river pollution surveys had done[2-4]. using a very simple river classification, the working party recommended a much more comprehensive river classification. This embodied both quality objectives and quality standards. These are shown in Table 1. A definitive policy document was subsequently issued by the National Water Council on this subject[5], and this was concurrently supported in a ministerial pronouncement by the Secretary of State for the Department of the Environment.

One more fundamental change on the question of river water quality which the working party highlighted was a recognition of the variation in surface water quality. This was accepted and allowed for by a consideration of class-limiting criteria that did not insist upon a satisfaction of all criteria for all determinands all of the time. The concept of the application of 95 percentile limits for quality criteria was advanced and accepted as a general policy. The reasoning was that it must be accepted as axiomatic that a specification of river quality requirements cannot be achieved for 100 per cent of the time. The only practicable alternative is to seek to specify the maximum levels of contamination of waters at a given statistical level of probability. Experience suggested that in general such maximum limits should not be exceeded for more than 5 per cent of the time.

The natural sequel to the above report was one on the effects of discharges to tidal waters and was also set up under the auspices of the National Water Council. A report was prepared at the request of directors of Scientific Services Group[6] and the Council has now agreed that the report should be circulated for use by water authorities.

The report provides a very substantial guideline on the environmental quality objectives that are pertinent to estuarial and coastal waters, (see Table 2). The working party was very conscious of the limited data that exist in general for tidal waters compared to freshwaters and this was reflected in the more limited statements on quality standards. Where a statement on quality standards could be made, then it was made, otherwise general quidelines only were indicated. Nevertheless, it was felt that, taking the nation's tidal waters as a whole, most tidal waters are substantially unpolluted and, accordingly, the primary objective of pollution control must be to prevent significant deterioration of the present quality.

It is expected that the water authorities themselves would determine the objectives for their own tidal waters. The main report gives some guidance on the quality standard relevant to each objective, whether in respect of E.E.C. Directive, now or soon to be in force, or any other relevant quality criteria. Thus, if specified standards (or criteria) were designated as operational objectives, it should then be possible to specify the limits of quality and quantity which could be applied to achieve the specified quality objective.

In contra-distinction to the report on the quality of freshwaters, it was felt to be generally inappropriate to express quality standards and hence consents to discharge on a 95 percentile basis. A decision on whether or not this is applicable should be left to the discretion of each water authority. The reasoning behind this was that frequency of sampling and sampling in general was very much less than that for freshwaters and that perhaps undue emphasis may be placed upon derived data that could not be substantiated on a sound statistical basis. Furthermore, there will be, nationwide, many discharges of effluent to tidal waters that will not even be quantified. However, in the case of the Thames estuary, very adequate sampling has been carried out for a very long period of time, even from the end of the last century, and therefore it is our policy to adopt a 95 percentile approach in respect of our quality standards and of the discharge consents that we are responsible for.

It was also felt by the working party that with some estuaries (the upper reaches of which are substantially freshwaters) may be treated in a similiar manner to freshwaters themselves. Similarly, the calculation of the consents to discharge may be done as if the effluent is discharging to a freshwater stream. The Thames Water Authority has adopted this approach in the tidal region from Teddington Weir to London Bridge.

SETTING OF ENVIRONMENTAL QUALITY OBJECTIVES

At the instigation of central government, all water authorities were required to set river quality objectives for all their significant freshwater streams and watercourses. To this end, the Thames Water Authority classified 2600 kilometres of its main freshwater rivers into appropriate river classifications according to the table of categories shown in Table 1.

This exercise was carried out in 1978, using the previous 3 years' river data, and a report[7] was presented to the Authority's members and subsequently to central government. In a later report[8], a further 1200 kilometres of rivers was also classified although these latter were, in general, smaller watercourses and insignificant streams into which there was little or no discharge.

This set the scene for the current situation in respect of river quality objectives, and these were referred to as our short-term objectives for which the minimum requirement would be to maintain that position, viz-à-viz quality. Further to this, our long-term objectives and aims were also specified in the same reports. Thus, the presentation of our rivers (or stretches of rivers) were defined either as e.g. River Mimram as class 1A or a stretch of the River Blackwater as class 3/2/1B. The first river class is self-explanatory in that its current river category is 1A and it will be maintained as such. The Blackwater river is currently in class 3 and as further improvements or more stringent discharge consents are invoked, it is anticipated that a river of class 1B will be ultimately attained, via an intermediate improvement to class 2.

The completely classified major river stretches are shown in Table 3 whereby our current objectives are shown as well as our intended long-term aims.

In saline waters, i.e. tidal and coastal, the situation is very much less clear, and the prospect of a similiar, systematic classification as for the freshwaters is impracticable at the present moment; and with our present knowledge. The major factors to be taken into account are the objectives outlined in Table 2, and there was additionally a Royal Commission report[9] which outlined two principal objectives related specifically to estuarine quality.

These were:

(a) the quality of water in an estuary should be capable of supporting the passage of migratory fish at all states of the tide, and
(b) the esturary should support on the mud bottom those organisms required to sustain sea fisheries.

As a result of these very general objectives the Thames Water Authority has set its quality objectives for the Thames estuary and these are shown in the accompanying Table 4 (refer also to Figure 3).

The objectives, as shown, and their accompanying quality standards are primarily based upon adequate concentrations of dissolved oxygen. In general, however, other standards may be required dependent upon the use to which any such waters are put. Indeed, the absence or low levels of chemical pollutants below levels toxic to biota or the human consumer, or causing tainting of fish flesh, may also be needed.

For bathing waters[10] and shell fisheries[11], microbiological standards may be required, in addition. Protection of human consumers of fish and shellfish may dictate a water quality standard more stringent than that required for the organism itself. Standards for persistent pollutants (heavy metals, pesticides, etc.) are difficult to derive. The toxicities of many such materials to sea fish and shellfish are known[12,13] but frequently the results are based on LC_{50} tests using unrealistically high concentrations for short duration under unrepresentative test conditions.[14]

Unfortunately, there are no guidelines in respect of fish toxicity similar to that of the EIFAC recommendations for freshwaters.[15] Standards for the Drins[16] (aldrin, dieldrin, and endrin) and mercury[17] are prepared in future E.E.C. Directives. A Directive on cadmium will follow also in due course. In the case of mercury, standards for saline waters (50 nanogram/litre) and fish flesh (0.3 mg/Kg) have been proposed. For other pollutants, some guidance can be obtained from the criteria published by the U.S. Environmental Protection Agency.[18] Some values are shown in Tables 5 and 6, although it should be stressed that the values are not intended as recommended standards. Criteria specify concentrations in water which, if not exceeded, are expected to result in "safe" conditions and are derived from laboratory and field tests usually short-term.

ACHIEVEMENT OF OBJECTIVES VIA DISCHARGE CONSENTS FOR FRESHWATERS

Although dishcharge consent conditions will remain as fixed figures which cannot legally be exceeded, it is not envisaged that with the full implementation of the 1974 Control of Pollution Act prosecutions will ensue, where only occasional samples fail to comply. In the light of experience in applying this, it is the Thames Water Authority's policy to adopt a scheme whereby 95 per cent of samples taken would have to comply with all conditions taken together. This would be the legal consent condition. However, consents determined on the basis of this procedure would be unsatisfactory for use as guidelines for management of sewage treatment. For this purpose, an additional new concept has been formulated called an operating target. The targets set for each works would be calculated on the basis that 95 per cent compliance with each determinand judged separately (be it suspended solids, biochemical oxygen demand or ammonia or whatever) would ensure that the present objective of the recipient waters would be maintained.

There are a number of ways[19] that these figures may be arrived at:

(a) ordered ranking
(b) simulated modelling
(c) simple mass balance calculation

The 'ordered ranking' procedure is the simplest to calculate and may be used when good reliable past data are available. The results for each determinand, separately, are ranked in ascending order of numerical values and the 95th percentile estimated and used as the operating target level.

The 'simulated modelling' method is particularly useful in circumstances where few data exist or hypothetical situations are to be considered, such as new works development or new discharges. The procedure assumes either a normal or log-normal distribution of flow and quality data and is based on the conventional mass balance equation whereby the sum of the loads upstream of a discharge point and the effluent discharge itself is equal to the load downstream of the discharge position. Data conforming to log-normal distributions are generated with defined means and standard deviations. These data are combined to give a resultant distribution of quality downstream of the discharge point. The required effluent quality is computed such that the downstream class limiting standard is exceeded for only 5 per cent of the time. This, of course, gives the 95 percentile value for the discharge condition.

The third method for mass balance is essentially a simpler version of the simulated modelling technique. It is more useful when very limited data exist or when an effluent is diluted little by a receiving stream.

As examples of the two latter ways in which a consent may be calculated, consider the following hypothetical situation, in conjunction with Figure 4, for the calculation of a consent based on BOD (ATU) to satisfy a downstream objective of class 1B.

Upstream conditions

Mean flow	9.7 m^3/s
5-Percentile flow	3.3 m^3/s
Mean concentration	1.3 mg/l
Standard deviation	0.7 mg/l

Effluent conditions

Mean effluent flow	0.8 m^3/s
Standard deviation	0.2 m^3/s
Current mean concentration	10.0 mg/l
Standard deviation	3.0 mg/l

Downstream conditons

Class limiting criterion	5.0 mg/l

The simulated modelling procedure, based on a computer program, gave a 95 percentile effluent consent value for the BOD (ATU) of 24.5 mg/l, associated with an average value of 15.8 mg/l. Clearly, those values would in practice be rounded off to 25 and 16 mg/l, respectively. A calculation using the simple mass balance method and the equation associated with the figure gave a 95 percentile value of 19.4 mg/l for the maximum permissible concentration BOD (ATU) in the effluent; this tends to be over-stringent, and gives only an approximation to the 95 percentile whereas the former method would produce a more accurate figure.

ACHIEVEMENT OF OBJECTIVES VIA DISCHARGE CONSENT FOR THE THAMES ESTUARY

In this case, it is the intention to discuss a specific application to one particular estuary (the Thames) rather than to attempt to generalize for all estuaries in U.K. waters.

When the pollution of the Thames tideway was at its most severe, during the 1950s, the Thames Survey Committee requested the Water Pollution Research Laboratory (now the Water Research Centre) to study this estuarial system. The result was the publication of a massive tome[20] and the production of a well-validated mathematical model,[21,22] the latter was taken over and used by the Thames Water Authority. A subsequent paper has described the use of this model for management purposes of the Thames estuary[23] for present-day purposes and for estimates in the future up to the end of the century.

The most critical period for the tideway is the third (summer) quarter when temperatures are high, thus increasing the rate of biochemical oxidation, and upland flows are low so that less water is available for dilution. Therefore, if standards for effluent quality are chosen to enable the quality objectives to be achieved under these 'worst' conditions, then the quality should be maintained through the remainder of the year. The procedure that was adopted was to use the mathematical model to determine whether the present average quality of effluents will achieve the desired quality objective of 30 percent dissolved oxygen saturation during the summer quarter. Table 7 shows the present average and 95 percentile values of effluent qualities for four major sewage treatment works in the middle reaches of the tideway. The application of these data by the model gave a minimum average dissolved oxygen saturation of 31 per cent for the summer quarter conditions. This is in very good agreement with the desired objective of a 30 per cent minimum dissolved oxygen value. Thus the 95 percentile figures in the table were selected as the operating targets for those works.

THE WAY AHEAD

It is quite clear that a wealth of data exists for freshwater rivers that enables statutory bodies to assess their situations in a detailed manner. There are probably as many ways of defining a river category as there are countries with a river classification. Whatever techniques are used, it is axiomatic that the use to which a river (or stretch of a river) is to be put must first be defined. This definition can be one that satisfies the current situation or indeed the pollution control authority can decide what the ultimate use of the river may be. This may be described as setting current objectives and, if required, future objectives. The quality criteria, or standards, are reasonably well documented for most objectives. Some determinands may be a little speculative and in these cases, clearly more work is required so that unnecessarily stringent (or relaxed for that matter) conditions are not introduced for the sake of setting a value. In the case of European countries, the application of EIFAC standards, E.E.C. Council Directives, International standards and guidelines, as well as each country's own standards are well-documented and should be used.

Thus, for freshwaters, the river quality objectives and standards can be quite readily ascribed with a fair degree of confidence.

In the case of estuarial and coastal waters, there is much to do inasmuch as there is a distinct lack of adequate guidelines on the choice and quantification of quality standards, only the quality objectives do not seem to present an obstacle. A great deal of research work is needed to be able to set adequate and well-founded standards. If the speed with which EQOs and EQSs (environmental quality standard) have been decided for mercury is anything to judge by, then many decades will elapse before other metal limits are arrived at.

Of course, there is the simplistic approach of waiting until an international obligation is foisted upon one so that countries must then simply comply. The snag with that is that all countries do not possess the same degree of honesty, and the well-meaning nation suffers. The other reason for not accepting that situation is that if a standard is forced upon one there is the underlying feeling that the level is not quite right. This infers that each nation should do its utmost to be able to substantiate a standard.

There is yet a further contentious area in particular between the U.K. and the rest of the E.E.C. This involves the application of quality standards to the achievement of environmental quality objectives. The U.K. view (or philosophy) is to treat each river or stretch of river on its own merits, i.e. to allow discharges into it that are compatible with what it can assimilate knowing the use to which it is put, and the dilution that is available to it. The Brussels arguments are in favour of fixed emission standards, the sole merit being or seeming to be that from an administrative viewpoint they are very easy to monitor. My own viewpoint is that the two philosophies may be considered as complementary, rather than mutually exclusive. The non-biodegradable substances with very low toxic levels could be ascribed fixed limits as levels of emission whilst degradable materials could be flexibly treated using the natural in-river purification.

Acknowledgement

The author is indebted to the Thames Water Authority and the Director of Scientific Services for permission to publish this paper.

Figure 1

Regional Water Authorities in England and Wales

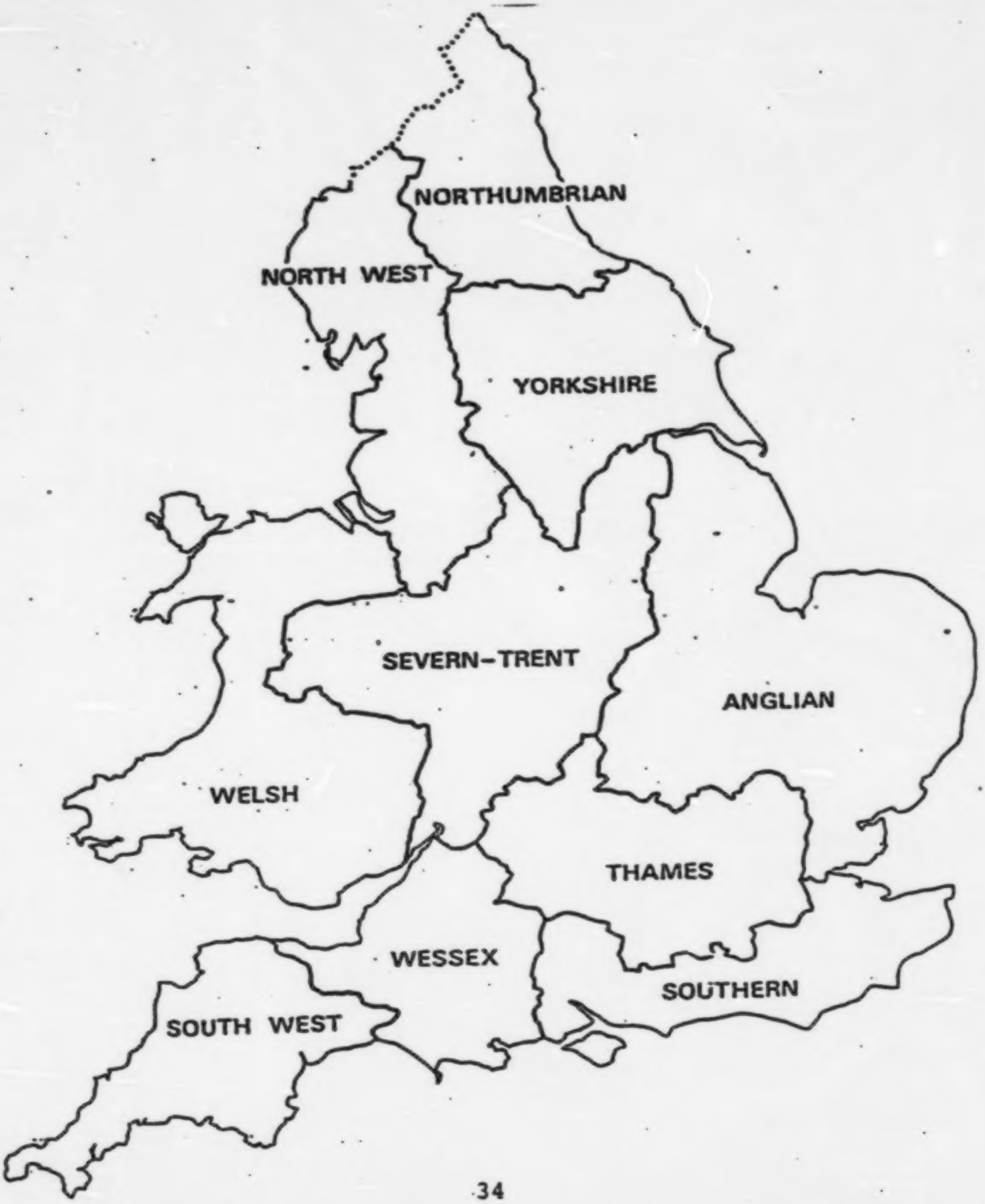


Figure 2

Organization of water management in England and Wales

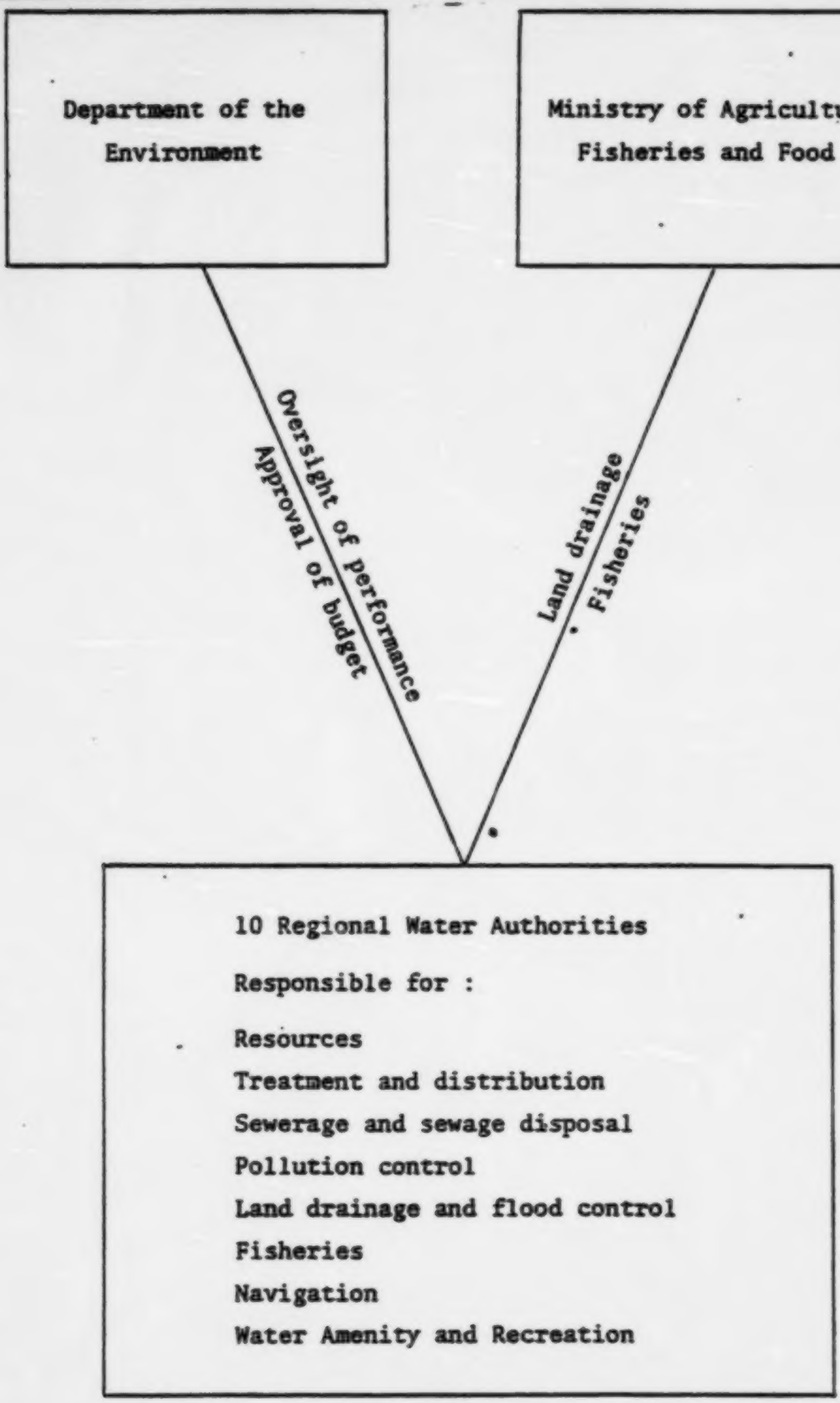
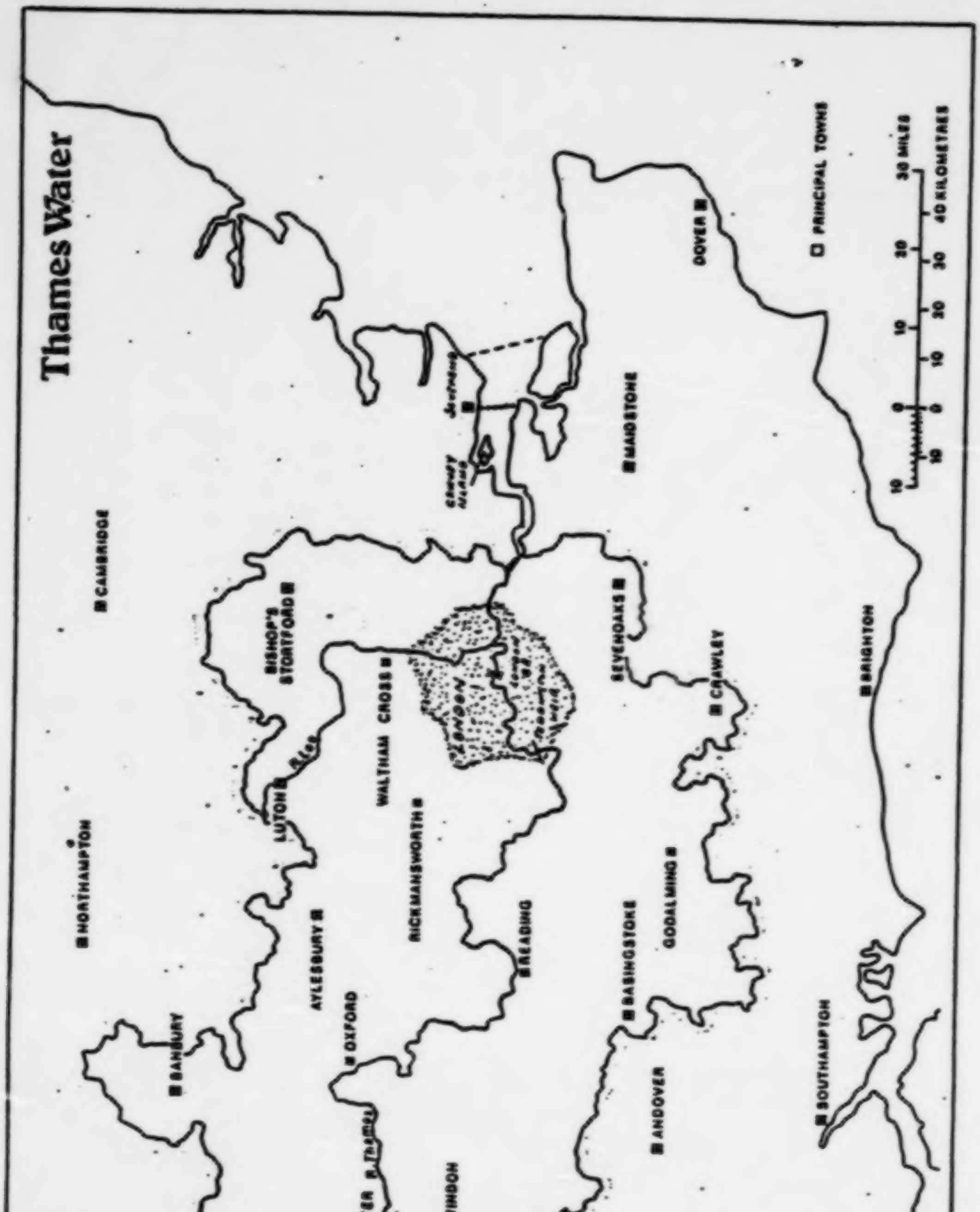

Department of the Environment
Ministry of Agriculture, Fisheries and Food
Oversight of performance
Approval of budget
Land drainage
Fisheries
10 Regional Water Authorities
Responsible for :
Resources
Treatment and distribution
Sewerage and sewage disposal
Pollution control
Land drainage and flood control
Fisheries
Navigation
Water Amenity and Recreation

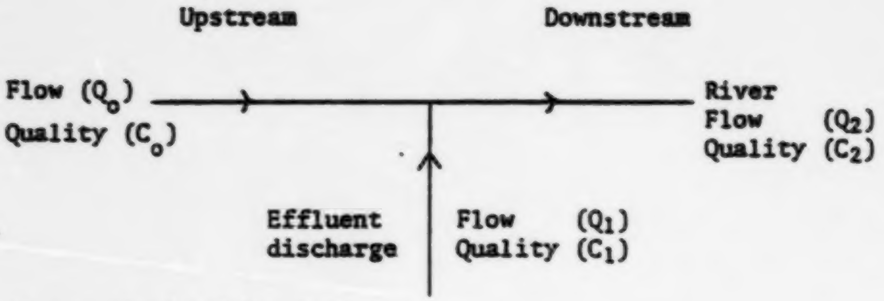
Figure 3



Thames Water
CHELTENHAM
CIRENCESTER
R. Thames
SWINDON
BANBURY
OXFORD
AYLESBURY
NORTHAMPTON
LUTON
R. Lea
RICKMANSWORTH
READING
WALTHAM CROSS
BISHOP'S STORTFORD
CAMBRIDGE
LONDON
Canvey Island
Southend
MAIDSTONE
SEVENOAKS
CRAWLEY
GODALMING
BASINGSTOKE
ANDOVER
SOUTHAMPTON
BRIGHTON
DOVER
PRINCIPAL TOWNS
10 0 10 20 30 MILES
10 0 10 20 30 40 KILOMETRES

Figure 4

Mass balance relationship for freshwater streams (used in simulated model calculations)



these relationships hold

$$Q_oC_o + Q_1C_1 = Q_2C_2$$

and

$$Q_o + Q_1 = Q_2$$

so that

$$(Q_o + Q_1)C_2 = Q_oC_o + Q_1C_1$$

$$C_1 = \frac{(Q_o + Q_1)C_2 - Q_oC_o}{Q_1}$$

where

Q_o ≡ 5 percentile flow upstream of the discharge.

C_o ≡ Mean concentration upstream of the discharge.

Q_1 ≡ Average effluent flow.

C_1 ≡ Maximum concentration permissible in the discharge (approximation to 95 percentile).

Q_2 ≡ Downstream river flow is the sum of the upstream flow and the discharge.

C_2 ≡ 95 percentile limiting concentration in the river set by classification.

Table 1

Classification of freshwater river quality and associated quality objectives and standards

River class	Quality Objectives	Quality Standards (95 percentile basis)
1A	(i) Water of high quality suitable for potable abstractions and for all other abstractions	(i) DO saturation greater than 80%
	(ii) Game and other high-class fisheries	(ii) BOD not greater than 3 mg/ℓ
	(iii) High amenity value	(iii) NH_3 not greater than 0.4 mg/ℓ
	(iv) Visible evidence of pollution should be absent	(iv) Where water is abstracted for drinking water, it complies with EEC - A2 category
		(v) Satisfies EIFAC requirements
1B	As for 1A, but cannot be placed in that category because of high proportion of high quality effluent present	(i) DO saturation greater than 60%
		(ii) BOD not greater than 5 mg/ℓ
		(iii) NH_3 not greater than 0.9 mg/ℓ
		(iv) As for 1A
		(v) As for 1A

Table 1 continued

Classification of freshwater river quality and associated quality objectives and standards

River class	Quality Objectives	Quality Standards (95 percentile basis)
2	(i) Waters suitable for potable supply after advanced treatment (ii) Supporting reasonably good coarse fisheries (iii) Moderate amenity value	(i) DO saturation greater than 40% (ii) BOD not greater than 9 mg/ℓ (iii) Where water is abstracted for drinking water, it complies with EEC - A3 category (iv) Satisfies EIFAC requirements or best estimate if EIFAC not available
3	(i) May be used for low grade industrial abstraction (ii) Waters that are polluted to an extent that fish are absent or only sporadically present (iii) Considerable potential for further use if cleaned up	(i) DO saturation greater than 10% and unlikely to be anaerobic (ii) BOD not greater than 17 mg/ℓ
4	Must be cleaned up since these are grossly polluted and are likely to cause nuisance	Inferior to class 3
X	Insignificant watercourses and ditches where the objective is simply to prevent nuisance developing	DO saturation greater than 10%

Table 2

Quality Objectives for tidal waters

Category	Quality Objectives
1	Protection of fishery resources including the reproduction, food supply, and habitats of the fish involved, for the maintenance of these resources and to ensure that where these resources are harvested, their quality is suitable for human consumption, although in some circumstances appropriate cleansing treatment may be considered necessary
2	Protection of other particular flora and fauna, and their habitats, for special conservation purposes
3	Protection of abstractions made for industrial, agricultural, or potable supply
4	Protection of bathing and other recreational uses of the waters
5	Avoidance of public nuisance

Table 3

Distribution of total river lengths among their various classes

Quality Class	Short-term objectives		Long-term objectives	
	Km.	% of total	Km.	% of total
1A	438.2	17.9	502.5	20.6
1B	952.5	39.0	1501.3	61.5
2	759.6	31.2	427.2	17.5
3	288.1	11.8	10.9	0.4
4	3.4	0.1	0.0	0.0

Table 4

Objectives and standards for the middle and lower reaches of the tideway

Stretch of tideway	Quality objectives	Quality standards
London Bridge to Canvey Island	(i) Freedom from putrescence and avoidance of public nuisance	The minimum air saturation with dissolved oxygen should be at least 10 per cent (95 percentile)
	(ii) Allow the passage of migratory fish	The quarterly minimum average percentage saturation with dissolved oxygen should not fall below 30 per cent, in the summer quarter
Canvey Island to seawards limit	The quality should be suitable for the whole life cycle of marine organisms including fish	The maximum air saturation with dissolved oxygen should be at least 60 per cent (95 percentile) during the summer quarter

Table 5

Water quality criteria published by the USEPA for North American fisheries

Substance	Levels which could constitute a hazard ($\mu g/\ell$) (ref. 18)	Levels at which there should be minimal risk ($\mu g/\ell$) (ref. 18)	Criteria for marine aquatic life ($\mu g/\ell$) (ref. 18)
Aluminium	1500	200	-
Arsenic	50	10	50
Cadmium	10	0.2	5
Chromium	100	50	100
Copper	50	10	0.1 of 96-h LC_{50}*
Iron	300	50	-
Lead	50	10	-
Manganese	100	20	100
Nickel	100	2	0.01 of 96-h LC_{50}*
Zinc	100	20	0.01 of 96-h LC_{50}*
Cyanide	10	5	5
Fluoride	1500	500	-
Sulphide	10	5	2†

* The 96-h LC_{50} test should use a sensitive resident species

† The criterion is 2 $\mu g/\ell$ of undissocated H_2S.

Table 6

Concentrations in water of some organic pollutants which could lead to tainting of fish flesh from ref. 18

	Threshold concentration
Substance	(µg/ℓ)
Phenol	1000-10,000
p-Cresol	120
Guaiacol	82
o-Chlorophenol	0.1-15
p-Chlorophenol	10-50
Dichlorophenol isomers	1-84
2, 4, 6-Trichlorophenol	3-50
o-Dichlorobenzene	250
n-Butylmercaptan	60
Pyridine	5000-28,000
Kerosene	100

Table 7

Consent conditions to be applied to sewage treatment works in the middle reaches of the tidal Thames

Sewage treatment works	Flow (m^{3}d/)	BOD (mg/ℓ)		Ammoniacal Nitrogen	
		Mean	95 percentile	Mean	95 percentile
Beckton	1,000,000	10	22	2.0	6
Crossness	568,000	12	25	7.5	16
Riverside	95,000	20	31	30	44
Long Reach	200,000	20	28	40	53

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DEVELOPMENT AND APPLICATION OF WATER QUALITY CRITERIA AND STANDARDS IN MEXICO

Eloy Urroz Jiménez

Eco-Ingeniería, S. A.
Mexico City
Mexico

DEVELOPMENT AND APPLICATION OF WATER QUALITY CRITERIA AND STANDARDS IN MEXICO

INTRODUCTION

It is worth while to present a brief summary of the hydraulic resources of Mexico. By knowing their distribution in the Republic and the water quality that exists we will realize how important is the pollution control of the rivers, lakes and seas of Mexico.

HYDRAULIC RESOURCES OF MEXICO

The Mexican United States are formed by 31 states and a Federal District. The total area of the country is of 1,967,183 Km^2 (759,300 square miles) and the total population is of approximately 70 million people.

The annual rate of growth of the Mexican population is 3.6%. The urban population (60% of the total) has a higher and constant annual rate of growth of 5.1% or three times as high as the rate of growth of the rural population which is only 1.5% per year; this difference in growth rates is reflected in the increment of the demand for urban services such as water supply and sewerage.

Mexican economic development, subject to a mixed economic system, is characterized by an annual rate of growth of the annual per capita income of approximately 6.5% having reached a value of 780 US dollars per capita at the present time.

Nevertheless, the biggest problem facing Mexico is the low level of income in a large part of the population, mainly in the rural sector, as a result of the uneven distribution of wealth and the general rate of growth of the population.

The annual rainfall for the country as a whole is of 717 mm and its distribution is very irregular in space and in time. There exists regions with an annual precipitation of 5000 mm (Chiapas) and other regions, like the northwestern deserts, where practically there is not any rain.

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*Former Director General, Water Pollution Control, Secretariat Agriculture and Water Resources, Mexico.

Generally it rains only during the summer season, from June to October. Climatically 63% of the Mexican territory has been classified as arid, 31% as semiarid, 5% as semihumid and 1% as humid.

Most of the Mexican rivers are of the intermittent type. The average annual runoff is of the order of 357 million cubic meters. The estimate of the ground water resource is of 40 million cubic meters. The irregular distribution of the resource is caused by the fact that 50% of the total annual rain falls in only 10% of the territory where only 7% of the total population lives.

In addition to the problem of the poor distribution of the hydraulic resources with respect to time and space, there exists the problem that the biggest concentration of population in Mexico is located precisely in the central heights and generally in big cities where the hydraulic resource is limited. The metropolitan area of Mexico City, which has a population of nearly 14 million people and is located at 2300 m above sea level, probably presents the world's most unfavorable condition for a water supply operation. Of the 400 million cubic meters of water currently available, it is estimated that only 156 million cubic meters are utilized at the present time. The most important water uses are: irrigation of crops, domestic and industrial.

Table 1 shows the approximate distribution of water uses for different purposes for the years 1970 and 2000. These figures were estimated taking into account that the area under irrigation is going to increase from 2.7 million hectares to 8.0 million hectares and that the population will require, in the future, 425 liters per capita per day of potable water for an estimated population of 135 million people.

MEXICAN WATER LEGISLATION

Probably, it is legislation which shapes the economic, social and technological development of each country. Mexico has, from the beginnings of the century, a constitution which established that the state has the rights over the soil, the water and the resources from the ground, and that the state will impose regulation over such rights that it considers adequate for the development of the country.

The Federal Water Law, therefore, declares the hydraulic resource as a national property and gives to the state the authority to assign water volumes to the municipalities and towns and to give water concession to private parties. The interesting point in all this is that the national characteristic of the water is never lost, no matter that the water has been used and it is in the sewers of a town.

Table 1.

APPROXIMATE DISTRIBUTION OF WATER USES IN THE YEARS 1970 AND 2000

Water use	Yearly average consumption (millions of cubic meters) 1970	2000
. Irrigation	38,400	104,400
. Domestic, municipal and industrial	8,800	20,600
. Fishing in lakes, coastal areas and in the continental platform	100,000	110,000
. Other uses	10,000	20,000
Total	156,000	226,000

In spite of the above facts it is not the Federal Water Law which regulates the control of pollution of the hydraulic resources, but, based in a 1971 modification to the constitution, a Federal Law to Prevent and Control Environmental Contamination was enacted. This law was fundamentally based upon the powers of the health authorities of the country because it was thought that the pollution problem was one of health. I believe that while the most important factor in this matter is health, the environmental control can not be limited to the health problems since before them appear real problems of welfare and of economy.

The law of control of environmental pollution and its regulations of 1973, establish then, the federal characteristic of the state governments and the local authorities can not take any action even if they realize that a town or an industry is discharging its wastewater effluents to the lake of the city.

Also, it is necessary to explain that, even though the Mexican constitution establishes the responsibility of the municipalities to supply drinking water and to provide the sewerage service to the town, this is not happening in practice. What really happens is that the Federal Government establishes two or three party programs to assist with funds the construction of these systems, but it is the Federal Government itself which controls the operation and maintenance of the water systems through the Joint Commissions of Water Supply and Sewerage.

In some (a few) cases the Federal Government has left this responsibility to the state governments, which establish their own state commissions of water supply and sewerage. In these cases it is the state commissions who operate and maintain the systems. Nevertheless, all these commissions continue strongly dependent on the Federal Government in a political sense and in some cases in the economic sense.

On the other hand, the Federal Government is responsible for oil development (through Pémex), for the sugar industry development (26 sugar mills), for the fertilizer and pesticide industry and it controls a large percentage of the petrochemical and chemical industries. This situation has created a situation in which private industry is waiting for the Federal Government itself to comply with the water pollution control standards. Since 1976 all industry must have installed wastewater treatment plants or have chosen for the municipalities to accept their industrial wastewater to be treated in the municipal plants.

Therefore, the result, after four years, is that the municipalities and with them the industry connected to their sewerage systems, have been unable to comply with the regulation, in the sense of installing wastewater treatment plants, due to the total lack of economic resources. Since the state owned and partially owned industries have not complied for similar reasons, with the exception of some oil industry installations, to this date there has not been an important construction activity of wastewater treatment plants.

ARE STANDARDS NEEDED IN MEXICO?

The process to establish criteria and standards for water quality appears when there is a need to establish goals and objectives in the control of the environment. This matter was amply discussed in Mexico during the period of time needed to prepare the law and its regulations of water and air.

It is assumed that at the moment of establishing the quality goals and their objectives and agreement is reached within different national interests such as health and economy. Unfortunately we had to deal with a series of difficulties to achieve a quantitative appraisal of health values able to compete with the economic interests. This situation is continually presenting itself in Mexico due to the enormous lack of awareness in the Mexican people of the value of health. Nevertheless, it was necessary to establish specific standards to elaborate the existing regulations.

The lack of research, the lack of collection of environmental data and of monitoring the effects that the different unhealthy environments in Mexico have over the health of man, drove the government commission in charge of the establishment of standards to take as a basis of consultation the water quality criteria of the USPHA, the WPCF and other organizations, and the standards of some European countries, of the World Health Organization and the ones published by the USSR.

Therefore, the work in a first stage consisted in conditioning the criteria and standards of these countries to Mexican conditions, the result was the tables No. 2, 3, 4, 5, 6 and 7 (for fresh, saline and coastal water) of the Mexican regulations.

A few years later, experience appears to indicate that the procedure is not the adequate for a developing country where there are not historic, cultural and legal conditions which allow the adoption of a system of standards for the control of water pollution.

In Mexico, where the population is of more than 70 million people, 46.3% are less than 15 years old, 10.3% finished primary education and 0.45% finished college education. As a result of these figures, the decisions which steer the course of the country are always made by the Federal Government where the scientific, technological, and economic resources are concentrated. This situation presents a highly peculiar case: the

TABLE 2
CLASSIFICATION OF FRESH WATERS

Class	Uses	PH	Temperature °C	D.O. mg/1	Coliform Organ. 100 ml	Oil & Grease mg/1	Dissolved Solids mg/1	Turbitity JTU	Color CPU	Taste & Odor	Nutrients N & P	Floatables	Toxic Substances
				minimum	maximum	maximum	maximum	maximum	maximum	maximum	maximum		
DA	Drinking water and food industry uses with disinfection only, contact recreation	6.5 to 8.5	Natural plus 2.5 (a)	4.0	200 (fecal) (b)	0.76	no more than 1000	10	20	absent	(c)	absent	(d)
DI	Water supply w/ conentional treatment (cogulation, sedimentation, filtration & disinfection) or industrial	6.0 to 9.0	Natural plus 2.5 (a)	4.0	1000 (fecal) (e)	1.0	no more than 1000	natural conditions	(f)	(g)	(c)	absent	(d)
DII	Water adequate for recreation, conservation of Flora Fauna and industrial uses	6.0 to 9.0	Natural plus 2.5 (a)	4.0	10,000 (total) avg. minimal, no value > 20,000	Absence of Visible Film	no more than 2000	natural conditions	natural conditions	natural conditions	(c)	absent	(d)
DIII	Water for agriculture and industry	6.0 to 9.0	Natural plus 2.5 (a)	3.2	1000 (j) open for cultivation use	Absence of Visible Film	(i)	natural conditions	natural conditions plus 10	—	(c)	absent	(d)
DIV	Water for industry except pharmicutical and food	5.0 to 9.5	—	3.2	—	—	---	---	—	—	—	—	(d)

NOTES FOR TABLE 2

(a) Maximum 30° except when caused by natural conditions. Measured on the surface outside the zone of mixing which will be determined by the characteristics of the discharge.

(b) No more than 10% of the total samples in a month (5 minimum), can be more than 2000 fecal coliforms.

(c) Should not exist in quantities which will cause agal blooms.

(d) The criteria for toxic substances is as follows:

Any toxic substance alone or in combination should be present in concentrations such that would render the water of the receiving body inadequate for the specific use designated.

Table 3 summarizes some of the toxic substances that in accordance with the available information are under study in different parts of the world. The values of substances shown in this table are not limiting and are subject to modifications in accordance with future technological advancement.

(e) No More than 10% of the total samples in a month may exceed 2000 fecal coliforms.

(f) No artifical color which is not coagulated in conventional treatment will be permitted.

(g) Removable for conventional treatment.

(h) Average of 2000 fecal coliforms per month, no sample shall be greater than 4000.

(i) Conductivity no greater than 2000 umohs/cm. If SER is greater than 6, the Department of Water Resources will set values in each case Baron is limited to 0.4 mg/l, for greater values the authority will set final limits.

(j) For irrigation of vegetables which are eaten raw or fruits which are in direct contact with soil.

TABLE 3
CLASSIFICATION OF FRESH WATERS

	Maximum Limit mg/l			
Classification	DA	DI	DII	DIII
Arsenic	0.05	0.05	1.00	5.00
Barium	1.00	1.00	5.00	—
Boron	1.00	1.00	—	2.0
Cadium	0.01	0.01	0.01	0.005
Cobalt	1.00	1.00	0.1	1.0
Hexavalent Chromium	0.05	0.05	0.1	5.00
Mercury	0.005	0.005	0.01	—
Lead	0.05	0.05	0.10	5.00
Selenium	0.01	0.01	0.05	0.05
Cynanide	0.20	0.20	0.02	—
Phenol	0.001	0.001	1.00	—
Methylene Blue Active Substances (detergents)	0.50	0.50	3.0	—
Chloroform Extractable substances	0.15	0.15	—	—
PESTICIDES				
Aldrin.	0.017	0.017		
Chlorodane	0.003	0.003		
D.D.T.	0.042	0.042		
Dieldrin	0.017	0.017		
Endrin	0.001	0.001		
Heptachlor	0.018	0.018		
Epoxy of Heptachlor	0.018	0.018		
Lindane	0.056	0.056		
Methatoxichlor	0.035	0.035		
Organophosphates	0.100	0.100		
Toxafing	0.005	0.005		
Total Herbicides	0.100	0.100		
RADIOACTIVE	PICOCURIES/L			
Beta	1.000	1.000	1.000	
Radio 226	3	3	3	
Strontium	10	10	10	

TABLE 4
CLASSIFICATION OF ESTUARINE WATERS

Class	Uses	(1) PH	(2) °C Temp.	(3) D.O. mg/l	(4) Coliforms MPN Organisms 100 ml	(5) Grease & Oil	(6) Dissolved Solids	(7) Turbidity (J.U.)	(8) Color Odor & Taste	(9) Nutrients Nitrogen & Phosporus	(10) Setterable Material	(11) Toxic Substances
			Max. Limit	Max. Limit	Max. Limit							
EI	Exploitation of shellfish for raw consumption All others.	6.5 to 8.5	N.C. +2.5 (a)	4.0	Average of 70	(d)	(e)	(f)	(g)	(h)	(i) absent	(j)
EII	Recreation (primary contact) all other uses except EI	6.5 to 8.5	N.C. +2.5 (a)	4.0	200 Fecal (b)	(d)	—	(f)	(g)	(h)	(i) absent	(j)
EIII	Fish exploitation and any other use except EI & EII	6.5 to 8.5	N.C. +2.5 (a)	4.0	10,000 * monthly average (c)	(d)	(e)	(f)	(g)	(h)	(i) absent	(j)
EIV	Navigation and any other use except EI EII & EIII	5.0 to 9.0	—	3.0	—	(d)	—	—	—	(h)	(i) absent	(j)

* The above values refer to those waters out of the mixing zones (k).
* N. C. - Natural Conditions.

NOTES FOR TABLE 4

(a) Measured on the surface in the zone of mixing.

(b) No more than 10% of the total monthly measurements (five minimum) may exceed 2000 fecal coliforms.

(c) No values should exceed 20,000 total coliforms.

(d) No oil or petroleum product should be discharged in quantities which:
   (1) can be detected as a visable film
   (2) can cause stains in fish and/or invertabrates, or
   (3) form deposits of oily mud on the coast, shoreline or bottom of receiving waters
   (4) that become toxic

(e) The geometry should not be changed in a drainage basin or entryway of fresh water which can cause permanent changes in the isohalines of ±10% of natural variation.

(f) The following limits will be applied:
   (1) natural conditions plus 5% if the natural turbidity is between 0 and 50 JTU.
   (2) natural conditions plus 10% if the natural turbidity is between 50 and 100 JTU.
   (3) Natural conditions plus 20% if the natural turbidity is greater than or equal to 100 JTU.

(g) No effluent should be discharged with these characteristics unless it has been demonstrated that it is not adverse to the aquatic flora and fauna, nor impedes best use of the receiving body.

(h) Should not exist in quantities which promote algal blooms.

(i) Any discharge susceptible to sedimentation or which can cause the consumption of oxygen, opacity, or interference with benthic organisms in their respiration or nutrition.

(j) The following criteria will be followed to assign as per table 5 the maximum permissible discharges:

It should be determined through bioassays the LD_{50} at 96 hours. Preferably bioassays with continuous flow using life stages most sensitive and the important ecological or enconomic species will be used with the following applicable factors:

1/100 for pesticides and metals
1/20 for sulfates
1/100 for all other toxic substances

(k) The mixing zone for each discharge shall be 1/3 the area and/or volume in the section considered. This shall be enlarged to 2/3 the area and/or volume as long as and when the characteristics of the discharge and the receiving body as well as the number of discharges located in the vicinity of the mixing zone permit it. In these cases they shall remain in the estuary a free zone of passage for migratory species not less than 1/3 the area or volume of the section considered.

TABLE 5
MAXIMUM VALUES OF TOXIC SUBSTANCES IN ESTUARIES

Arsenic	1.00 as AS
Cadmium	0.01
Copper	0.05
Hexavalent Chromium	0.01
Mercury	0.005
Lead	0.1
Phenols	0.1
Methylene Blue Active Substances (detergents)	0.5
Nickle	0.1
Zinc	10
Cyanide	0.02
Sulfates	0.5
Florides	1.5
Ammonia	0.8
Cresoles	1.5

Pesticides

Aldrin	0.0004
BHC	0.02
Clordane	0.02
Endrin	0.002
Heptaclor	0.002
Lindane	0.002
D.D.T.	0.006
Dieldrin	0.003
Endosulfan	0.002
Methetoxichlor	0.04
Perthane	0.03
TDE	0.03
Toxefine	0.03
Coumaphos	0.02
Dursban	0.03
Phenithion	0.0003
Naled	0.03
Paration	0.01
Ronnel	0.05
Arsenates	0.01
Naturales	0.10
Carbamates	0.10
Derivitives of 2, 4 - D	0.10
Derivitives of 2, 4, 5 - T	0.10
Derivitives of Triazine	0.10
Derivitives of Urea	0.10

TABLE 6
CLASSIFICATION OF COASTAL WATERS

Class	Uses	(1) pH	(2) Temp. °C	(3) D.O. mg/l	(4) Coliforms MPN Organisms 100 ml	(5) Grease & Oil	(6) Transparency	(7) Color, Odor & Taste	(8) Floatable Matter	(9) Toxic Substances
				Max. Limit						
C1	Shellfish growth for direct consumption and acuaculture areas and all other uses	N.C. ±0.3	N.C. + 10% (b)	90% of N.C. (c)	The mean concentration should be ≤ to (f)	(j)	(k)	(m)	absence	(n)
C2	Recreation with primary contact. All other uses except C1	N.C. ±0.3	N.C. + 10% (b)	90% of N.C. (c)	Less than 1000 (g)	(j)	(k)	(m)	absence	(n)
C3	Recreational uses w/out primary contact. All other uses except C1 & C2	N.C. ±0.4	N.C. + 10% (b)	90% of N.C. (c)	Less then 2000 (h)	(j)	(l)	(m)	absence	(n)
C4	Fish exploitation of scaly fish. All other uses except C1, C2 and C3	N.C. ±0.4	N.C. + 10% (b)	90% of N.C. (c)	The monthly mean concentration should be ≤ 10,000 (i)	(j)	(l)	(m)	absence	(n)

* All these characteristics should be obtained from representative samples of the affected zone by the wastewaters, outside of the initial mixing zone (o).

NOTES FOR TABLE 6

(a) For coastal waters the mixing zone will be considered to be the volume adjacent to the discharge site in which the effluent mixes with coastal waters due to the momentum of discharge and density differences.

(b) Can never exceed 32°C.

(c) Should never be less than 4.0 mg/l.

(d) Should never be less than 3.0 mg/l.

(e) Should never be less than 5.0 mg/l.

(f) No more than 10% of the samples in a month may exceed 230/100ml.

(g) No more than 20% of the samples (at least five) may exceed 1000/100ml; nor any single sample taken during the period of verification of 48 hours exceed 10,000/100ml.

(h) No more than 20% of the samples should exceed the value considered in a monthly period, nor in a confirmation period of 48 hours can exceed 10,000/100ml.

(i) No more than 20% of the total samples can exceed 10,000/100ml in a monthly period, nor any exceed 20,000/100ml.

(j) Same as letter (d) notes for Table 4.

(k) The monthly average of this parameter cannot be reduced by one standard deviation of the average determined for natural levels.

(l) The monthly average of this parameter cannot be reduced by more than ½ standard deviation of the average determined for natural levels.

(m) No effluent should be discharged with these characteristics unless it has been demonstrated that it is not of aquatic life, physical appearance or the best use of the receiving water.

(n) The following criteria will be followed to assign according to Table 7 the maximum concentrations permitted in the discharge:

It should be determined by bioassay the LD_{50} (96 hours). It is preferred that the bioassays be made under continuous flow using the most sensitive life stages of the most important ecological or economic species, with the following application factors:

1/20 for all toxic substances

When it is not possible to determine the LD_{50} the average should be calculated by:

$$LD_{50}(96\text{ hrs.}) = \frac{170}{\log(100 - S)}$$

where S = % survival using 100% of the waste.

TABLE 7
CLASSIFICATION OF COASTAL WATERS

Maximum Permissible Values of Toxic Substances In Coastal Waters

Arsenic	0.1 as As mg/l
Cadmium	0.001
Copper	0.005
Hexavalent Chromium	0.001
Mercury	0.0005
Lead	0.001
Phenols	0.001

government legislates and regulates the control of water pollution only for itself.

I must clarify that the authorization of all public investments, as to the Departments at cabinet level, the state owned and part owned industry and the derivation of monies to the state governments is authorized by the Ministry of Program and Budget which is an organism of the Federal Government in charge of the total planning of the country.

Even though the Ministry of Agriculture and Hydraulic Resources, which is the agency in charge of enforcing the law and the regulations in the case of water, is on the verge of initating a program to establish committees that will allow the analysis of the parameters already established and to reach some conclusions over criteria and eventually to standards. I believe that is is not going to be necessary since the experience of six years of application of the regulation shows that the road should not be the one of specific standards of water quality. I believe that we are fast reaching a day in which the decision will be: or we accept a lot, a little or some pollution, or we simply do not accept any pollution. The reason for this is the process of change that society experiments in the value of the renewable and flux natural resources, simply it would have to use a technology able to be managed in the country.

This decision leaves at the side all standards currently in effect. Of course it would require a very careful sanitary control so that in a given time any discharge of wastewater be prohibited.

THE IMPORTANCE OF WATER REUSE

Considering the ecological conditions of Mexico, where almost 70% of the country is arid or semiarid, water reuse should strongly pattern the management of liquid wastes. It is expected that in the year 2000 Mexico will have 135 million people and this means that the limiting resource for life and development will be water.

I am sure that all the water that can be reutilized in a direct way will be used in industry, in agriculture and eventually in the municipalities. This is a reality in all the good of the country.

It is easy to understand that the criteria and the standards should take a totally different approach.

Rivers will be ever carrying less water, they will have to have more and more dams and a point will be reached where water consumption will be extremely high. At the present time up to eight wastewater treatment plants intended for reuse are being built in Mexico.

As an example I will mention the Valley of Mexico case where 14 million people live and 40% of the total industry of the country is installed. The Ministry of Agriculture and Water Resources has a program to construct six wastewater reuse plants for a total flow of eight cubic meters per second.

APPROACH FOR THE FUTURE

Taking into account the above mentioned concepts I believe that the effort should be directed to three specific areas:

1. Water quality control in estuaries and coastal waters.
2. The use of best available and applicable technology for reservoirs, lakes and aquifers.
3. Strict control of standards and operation of plants for water reuse.

Of these three areas, the last appears to be the one that more attention should be provided and within this area, the sanitary and health aspects should have priority.

CONCLUSIONS

On studying and participationg in the studies and programs of water pollution control in countries with humid ecologies, it is necessary to protect water quality in a platform of trade offs between the different beneficial uses of water and the economic interests of enterprises and municipalities.

In the case of countries with little water or with extreme hydrological regimes, as is the case of Mexico, the approach should be outlined from its conception in another way:

1. The planning of use and reuse.
2. Application of the most advance operating technology in the arid and semiarid regions of Mexico.
3. Control of conservative pollutants in seas and estuaries.
4. Establishment of zero discharge in lakes, ponds, reservoirs, tourist zones, recreational zones, border zones, ground water aquifer and drinking water supply sources.

CRITERIA AND STANDARDS:
WHICH IS WHICH

Nancy Othmer

National Resources Defense Council
122 East 42nd Street
New York, N. Y. 10017

Criteria and Standards:
Which is Which

When I was asked to talk about the difference between water quality standards and water quality criteria I figured I'd have the shortest paper ever given. A single sentence, I thought: water quality criteria are not standards but they can be incorporated into standards through a legal process involving a hearing and opportunity to comment.

Well I guess a little more must be said. After all, the confusion in people's speech and in some written documents too, is awesomely persistent.

Let's start with criteria, the specific 65 Criteria Documents EPA is now preparing for final publication this fall. These criteria documents are being prepared to comply with the Consent Decree in NRDC v. Train issued in June 1976, almost exactly four years ago. EPA spent nearly two years working with an outside contractor before it set forth on the effort that is producing the documents you have seen in draft form over the last year. As currently being developed the documents will recommend maximum permissible concentrations consistent with the protection of aquatic life, human health and recreation. Thus at least in draft form they provided concentrations, numbers, representing scientists' views of how much of pollutant A or B could be in the water and still have a viable aquatic ecosystem. To protect human health and recreation, they provided numbers thought to be protective against various identified effects, including, where appropriate, numbers identifying the probable increase in the incidence of cancer from various pollutant loadings.

As I understand it now, EPA has modified its approach to both kinds of numbers, and will be issuing only a third or a half of its documents with complete sets of numbers. Other pollutants will be addressed with various, generally less satisfactory methods, of derivation, such as the application factor approach which should be familiar from earlier editions of criteria like the Redbook. Still, we can expect a sizeable number of pollutants to have pretty well-researched concentration levels as well as other numbers with less formal but still interesting numbers.

These documents are being published under the authority of §304 - the "guidelines" section for both technology-based best practical treatment (BPT) and best available treatment (BAT) of §301, as well as for the §303 water quality standards program.

Let me give a quick sketch of the water quality standards program as it is structured in the Clean Water Act.

Before 1972 water quality standards were the major basis on which water pollution was controlled - if it was at all. States were required under a 1965 Act to establish standards for all interstate waters, and authorized, I believe, to set standards for waters solely within their boundaries.

By the early 1970's when Congress was planning to amend the '65 Act it had become clear that the existing standards program was close to useless. First of all, much of the program was voluntary. Second, the standards were devilishly hard to enforce. How do you hold company X responsible for pollution excesses miles downstream? Various states did set up standards programs and did try to make them work. But the 1972 amendments took a bold brush and revamped the whole program.

The major new thrust of the 1972 Act was the effluent guidelines limitation program. Some 20 major industrial groups have been regulated under this program. The first round of technology-based limitations, imposed on individual dischargers, is in effect now. These limits are based on the Best Practicable Treatment (BPT) for industry, and secondary treatment for municipal treatment plants. Now the agency is coming out with its second stage controls, BAT or "Best Available Treatment". Theoretically these controls should be in effect by mid-1984. There is, I think, a considerable doubt about whether that date will be met. So now the water quality standards program acquires a newfound urgency and purpose. People are anxious that - or lest - the standards program EPA announced very tentatively in mid-1978 will step in where BAT is inadequate or simply non-existent.

Let me give a quick runthrough about water quality standards programs. These are basically state programs. States select and define the uses they think appropriate for their waters. High quality for water for swimming, recreation, fishing, drinking - lower quality for things like agriculture, industry, whatever. Some states just call them classes A, B and C. States were asked in the first round of standard setting to aim for the higher quality uses as it was these uses that Congress intended all our waterways to achieve by 1983. Congress also, in section 303, directed that water quality standards were to enhance the quality of water and serve the purposes of the Act.

Water quality standards are the initial job of the states. Under section 303 states are to review and revise their standards program every three years. The results of these reviews are to be submitted to EPA. EPA must then approve the standards or disapprove them. If it disapproves a standard EPA can then work with the states to develop satisfactory standards or if that fails, propose and promulgate new or revised standards of its own. So far as I know there have been only a small number of such actions - most of them quite successful. The point to be made, I think, is that water quality standard programs really are critters of the state rulemaking process. But the Act puts a big stick in the hands of EPA so that if states are not able to put together strong and effective programs, EPA has full authority to set up standards that will do the job.

Perhaps the major difference between water quality standards and criteria is that one has regulatory effect and the other does not. A criterion may be used in a standard which has mandatory effect, but is not in itself a regulation.

Now what are these standards and how do they differ from criteria? A typical state standard program will have at least two different uses. The higher quality use let's call fishable/swimmable. Take your favorite lake or stream, say Lake George in the Adirondacks, or Tahoe out west. If you can still fish and swim in them, they're likely to be pretty clean. Take the lower Hudson River - I don't know if swimming is allowed there, but I'm pretty sure I would swim very carefully if I found myself in it. In the theoretical scheme I've drawn here, the lower Hudson would probably be classed for industrial uses.

Now each of a state's uses is associated with a number of pollutant concentration limits. An industrial use might allow a pH of 4 or 5, while fishable/swimmable Lake George would require pH of between 6 and 9. Likewise the less clean use might permit higher concentrations of BOD, tetrachlorethelene, heavy metals, or ammonia.

What people have been discovering as they've checked into the quality of different waterways is that a great many waters are contaminated with far larger amounts of toxic pollutants than had been expected.

Thus a standard is comprised of a use designation and a series of numbers or concentration limits. Unfortunately, and this may be the cause of some of the confusion, these numbers are frequently called criteria. In fact, of course, the word criteria would make sense for describing the number parts of a standard, if it weren't such a source of confusion. For clarity, I prefer the phrase concentration limit.

But a criterion that has become part of a standard is not the same as criteria published by EPA under §304 and the Consent Decree. What EPA publishes has no force of law. Its criteria are merely recommended concentration limits for the protection of specified uses. To be adopted as the number component of a standard to acquire a legal effect, these EPA "criteria" must go through a state regulatory process.

Without this process or a federal rulemaking process, the "criterion" can have no legal force.

Now aside from the semantic and legal confusion there is another sense in which EPA's criteria documents are being mislabeled standards. This confusion is rooted in an advanced notice of rulemaking that came out about the time EPA published its first draft method for calculating criteria for the protection of aquatic life back in 1978.

EPA explained at that time that it was using the existing criteria (the Redbook ones) to judge state standards presented to it for approval. If the states justified different numbers, o.k., but if not, EPA was assuming that the Redbook numbers were the appropriate ones for specified higher uses. Numerous comments challenged this "presumptive applicability" - partly because of fear that these numbers would entail stricter effluent limitations and partly from a belief that the single set of numbers was inappropriate.

Consequently, when the first round of criteria documents was published a great deal of effort went into explaining how the new numbers were derived and how they were, as published, only numbers.

It's been about two years since these first intimations about criteria and standards were published. EPA now promises to publish final criteria documents this fall, and at the same time to propose a new water quality standards program to guide states in their standard setting and to announce and take comment on how EPA itself plans to use the criteria where states are reluctant or unable to do so.

Since the criteria, as I've said, are going to have various degrees of certainty, and various kinds of data bases, I think we can expect a pretty interesting proposal. You are all probably as familiar as I am with EPA's thoughts on the matter, but I'd be happy to try to answer any questions you may have.

WATER QUALITY CRITERIA AND STANDARDS--
A CONFLICT BETWEEN SCIENCE AND POLICY

Geraldine V. Cox

Chemical Manufacturers Association
1825 Connecticut Ave., N.W.
Washington, D. C. 20009

ABSTRACT

The Clean Water Act of 1972 (the Act) and its 1977 and 1978 amendments require states to adopt water quality standards which must be approved by the Environmental Protection Agency. The Environmental Protection Agency is required to develop water quality criteria as guidelines to the states to assist them in developing individual state water quality standards.

The conflict between science and government policy is based on the assumption that uniform water quality criteria can be applied across the country and that the discipline of aquatic sciences is sufficiently developed to be used for criteria establishment. The government is trying to regulate in a scientific area with an inadequate data base and hardly any laboratory-to-field confirmation of the existing laboratory data.

The Environmental Protection Agency should not publish criteria which cannot be supported by strong science and they should involve the total interested scientific community in criteria development and peer review.

INTRODUCTION - THE LAW

The objective of the Clean Water Act is ". . . to restore and maintain the chemical, physical and biological integrity of the Nation's waters." Furthermore:

> " . . . it is the national goal that discharge of pollutants into the navigable waters be limited by 1985;
>
> "it is the national goal that wherever attainable, the interim goal of water quality which provides for the protection and propagation of fish, shellfish and wildlife and provides for recreation in and on the water be achieved by July 1, 1983;
>
> "it is the national policy that the discharge of toxic pollutants in toxic amounts be prohibited; . . . "

Reading Section 304(a)(1) of the Act, the EPA Administrator ". . . after consultation with appropriate Federal and State agencies and other interested persons, shall develop and publish, within one year after enactment . . . (and from time to time thereafter revise) criteria for water quality accurately reflecting the latest scientific knowledge (A) on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, esthetics and recreation which may be expected from the presence of pollutants in any body of water, including ground water; (B) on the concentration and dispersal of pollutants, or their byproducts, through biological, physical, and chemical processes; and (C) on the effects of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication and rates of organic and inorganic sedimentation for varying types of receiving waters."

In Sec. 304(a)(2) the Administrator is required to publish ". . . information (A) on the factors necessary to restore and maintain the chemical, physical and biological integrity of all navigable waters, ground waters, waters of the contiguous zone, and the oceans; (B) the factors necessary for the protection and propagation of shellfish, fish and wildlife for classes and categories of receiving waters and to allow recreational activities in and on the water; and (C) the measurement and classification of water quality; and (D) . . . the identification of pollutants suitable for maximum daily load measurement correlated with the achievement of water quality objectives."

In Sec. 304(a)(4) the Administrator must ". . . publish and revise appropriate information identifying conventional pollutants, including but not limited to, pollutants classified as biological oxygen demanding, suspended solids, fecal coliform, and pH . . ."

In Sec. 304(a)(5)(A) the Administrator must ". . . develop and publish information on the factors necessary for the protection of public water supplies, and the protection and propagation of a balanced population of shellfish, fish and wildlife and to allow recreational activities in and on the water." The section says that these guidelines should be published within six months of the enactment of the 1977 amendments and they are to be used in consideration of Sec. 301(g) waivers from best practicable technology. These waivers can be granted when an owner of a point source can show that (Sec. 301(g)(c))

> "Such modification will not interfere with the attainment or maintenance of that water quality which will assure protection of public water supplies, and the protection and propagation of a balanced population of shellfish, fish and wildlife, and allow recreational activities, in and on the water and such modification will not result in the discharge of pollutants in quantities which may be reasonably anticipated to pose an unacceptable risk to human health or the environment because of bioaccumulation, persistency in the environment, acute toxicity, chronic toxicity (including carcinogenicity, mutagenicity or teratogenicity), or synergistic propensities."

In Sec. 304(a)(5)(B) the Administrator has six months to publish information for criteria which allows modification in discharge of any pollutant from an existing discharge from a publicly owned treatment works into marine waters provided that: "(1) there is an applicable water quality standard specific to the standard for which the modification is requested, which has been identified under Sec. 304(a)(6) of this Act.

In NRDC v. Train, 8 ERC 2120 (DDC 1976), EPA entered into a consent decree to settle a citizen's suit by an environmental group challenging EPA's failure to list and set standards for several hundred toxic pollutants. EPA apparently had difficulty in administering Section 307(a) of the Clean Water Act because it lacked flexibility to consider adequately costs and technological constraints and provided very short time frames for compliance. The consent decree provided a time schedule for the Agency regulating more toxic substances under the BAT effluent standards rather than solely under Section 307(a). The provisions of the consent decree, with only minor modifications, were incorporated into the 1977 Amendments of the Clean Water Act.

While all of these sections of the Clean Water Act seem redundant and somewhat confusing to aquatic scientists, it is important to understand the requirements and goals pertaining to the Act before considering water quality criteria and standards.

Criteria are nonbinding guidelines developed by the U.S. Environmental Protection Agency in consultation with state agencies and interested parties.

The Criteria should:

- be reflective of scientific knowledge for protection of aquatic biota, shorelines, beaches; and esthetics and recreation.
- consider biological, physical and chemical properties;
- reflect the knowledge of the effects of pollutants on diversity, productivity, stability and eutrophication on biological communities and the effects of organic and inorganic sedimentation for different types of waters.
- provide guidelines for waivers from point source discharge requirements to conform to the previously mentioned information plus prove that quantities of dicharged materials will not impose unreasonable risk to human health or the environment because of bioaccumulation, persistency in the environment, acute toxicity, chronic toxicity (including carcinogenicity, mutagenicity or teratogenicity) or synergistic propensities.

Standards are binding water quality measures promulgated by the states and approved by the United States Environmental Protection Agency. If the state does not promulgate standards or the EPA disapproves the state standards, the EPA can promulgate a substitute standard as the state standard.

On July 10, 1978, EPA issued an advance notice of proposed rulemaking setting forth a proposed regulatory program for the development of water quality standards (43 *Fed*. *Reg*. 29588). The published policy is that the fishable and swimmable water category is the national norm and state standards must be developed on this basis.

The standards are broken into use designation categories with A (the best) to D (the worst) for fresh water and SA (the best) to SD (the worst) for salt water. Each section of a state's water is to be classified by these use categories and a designation below class B (fishable, swimmable) must be substantiated to EPA by the state.

Background - the History of EPA's Criteria

The first guide for water quality criteria was the 1968 "Green Book". In 1974 the "Blue Book" was published by EPA. Both of these early documents were produced by committees which included the scientific community at large, state, and other public groups -- as the law mandates. These documents were accepted by the scientific community.

In 1976 EPA published the next edition of water quality criteria, the "Red Book". This was developed by an EPA contractor without benefit of ". . . consultation of appropriate Federal and State agencies and other interested persons . . . " as required by the law. This document was not accepted by the scientific community and a highly critical review was prepared by the American Fisheries Society[1]. EPA's own Science Advisory Board rejected the Red Book and fell back to the 1974 Blue Book as the last valid work. Yet, EPA uses the Red Book for some regulatory programs.

Since 1976, EPA has made several attempts to develop water quality criteria which were unsuccessful.

For the first set of toxic criteria that the Agency developed beginning in 1976, EPA staff, supplemented with summer students, performed literature searches on the 65 chemicals (and classes of chemicals) identified by the *Natural Resources Defense Council, et al. vs. Train consent decree*[2]. They recognized the extremely poor data base on these chemicals and the inaccuracy of the published literature to scientifically support the draft proposed criteria of 1977-78. EPA's Duluth Research Laboratory then published a complicated and inaccurate scheme to manufacture data when the data base was weak. The proposed calculation was published in the Federal Register in 1979. (Only a few of the 65 chemicals have a scientifically valid data base.) Rather than not issue criteria when the data were weak, the Agency tried to invent answers. This approach was a tremendous failure. The statisticians showed a 161% error for some calculations[3] and EPA was soundly, and justifiably criticized for this totally invalid approach.

Nevertheless, they pressed on and issued proposed criteria in 1979 using these calculations to develop criteria levels, although EPA corrected some mathematical errors in this data synthesizing method.

EPA plans to issue new criteria as final in 1980 using new guidelines which have not been published for public review on the premise that the public reviewed the last set and which the Agency did not approve and EPA now does not have time for another review. To my nonlegal mind that appears to be a clear violation of the letter and intent of the law requiring consultation with interested parties.

EPA's Science Advisory Board has copies of the newest document and the Agency may argue that this is public review, but it is not since states and interested parties were not allowed the opportunity to comment!

Science and Policy -- The Conflict

Nature is no respecter of man's attempts to classify and compartmentalize water bodies, plants, animals, and their ecology. Nature simply does not comply with Classes A to D waters and we should try to incorporate nature's variation into the Water Quality Standards.

Waters vary naturally from watershed to watershed within a state, let alone for the nation. The absurd presumption that a glacial lake in Alaska can be regulated by the same criteria as a Georgia fishing hole, as the Great Salt Lake, as Lake Michigan is unrealistic and naive. States must be allowed to develop realistic standards for their waters dependent on their natural conditions. For example, some natural streams have a healthy ecosystem under conditions that EPA has designated as toxic. Should we destroy a healthy, natural ecosystem to comply with government standards? I should hope not! States must be allowed to decide if different uses other than those of the National Policy are appropriate to achieve the objectives of the state.

Scientists are expected to know everything. We don't. The law stresses maintenance and restoration of diversity, productivity, stability and protection of fish, shellfish, plankton, etc. Yet EPA's post-1976 attempts don't even address this. The later criteria are focused on laboratory data on individual organisms. There is no requirement in the other sections of the Act for toxicity, carcinogenicity and other related data except in the guidelines for point-source waivers. The Agency does not consider the ecosystem at all in its criteria. Even the simplest ecosystems are not understood sufficiently well to be regulated. An ecosystem is dynamic and in constant flux depending upon physical, chemical and biological changes. A hurricane can cause damage to oyster beds which lasts for up to 10 years. An especially cold winter or warm summer can have significant effects on ecosystems. Fluctuations from parasites, viruses, etc., can alter populations. The recent eruption at Mt. St. Helens illustrates the effect nature has on an ecosystem. For the environment on and around the mountain to return to its pre-May 18 condition will take several generations. Mt. St. Helens has most affected the southwest corner of Washington State but has also affected the entire state, portions of all northwestern states, the U.S. and possibly the entire globe with climatic change. Mt. St. Helens in a few minutes introduced into the air more particulates than have been emitted by all of industry in the last 100 years. Consequently, to remove natural from man-induced variation is almost impossible except for grossly polluted environments. We don't have the ecological predictive capacity we need to regulate the environment.

The Agency is required to publish information on chemical, physical and biological factors and organic and inorganic sedimentation. The Agency published draft criteria that did not consider these factors. For example, EPA ignores the most fundamental chemistry such as chemical speciation, e.g., ionic state, chelation. They ignore the physical properties such as adsorption. EPA tests with the most toxic form of the study material and assumes that all of the material is in that form, e.g., heavy metals. Yet silver, chromium, copper and lead have very different toxicities based on their valence state. They ignore critical parameters such as water hardness which have demonstrated significance on toxicity for materials such as cadmium.

The conflict is easy to express but difficult to resolve. The agency lawyers want an easily measured and easily enforced set of numbers which can be used for regulatory purposes. The scientists don't have sufficient data to develop accurate numbers for ecosystem protection -- and probably never will.

The state of aquatic toxicology is primitive at best. When compared with mammalian toxicology, it is at least a generation behind . . . and the mammalian toxicologists generally consider themselves primitive vis-à-vis extrapolating data from animals to predictions on the effects on man.

Generally good data to regulate toxics are insufficent; hence, the Agency should focus only on the conventional parameters of biological oxygen-demanding materials, suspended solids, pH, fecal coliforms, etc.

Since the bulk of this nation's pollutants enter the water from nonpoint sources, point-source waivers under Section 301(g) should consider the current conditions of the receiving waters and whether additional treatment will substantially reduce that pollutant in the receiving water to a level that is justified by the treatment cost. Since this is the *only* section that discusses chronic and acute toxicity, the Agency should set *guidelines* for this site-specific determination and *not* try to develop criteria for every chemical. This would allow valid consideration of the individual site conditions.

The Agency *must* follow the Clean Water Act and develop the water quality criteria with other federal and state agencies and the interested public. Using comments on obsolete criteria published in the *Federal Register* is inadequate consultation for a final promulgation. The Agency may not feel that criteria are regulations, yet they have the impact of regulations. Therefore, they should be subject to the same review. A group of interested parties should *work together* to develop a good product. The Agency's adversarial attitude and isolationist policies lead to delays and frequently an inadequate work product. The government does not have the corner on all creative thought on water quality criteria -- the more people who input, the more comprehensive and the better the product.

With the guidance of interested parties, the EPA should embark on a meaningful program to develop an adequate data base for water quality criteria.

RECOMMENDATIONS

Water quality criteria can be established for such parameters as fecal coliforms and biological oxygen-demanding materials, and these can be segregated in State Water Quality Standards into the use categories of A through D.

Guidelines, not criteria, for state water quality standards for toxic material evaluation should be developed for state water quality standards. These guidelines should include discussions on the influence of:

- water hardness consideration on ecosystem toxicity;
- pH effects on ecosystem toxicity;
- chemical/physical interaction as they affect aquatic community toxicity;
- the persistence of chemicals and their long-term effects;
- the natural communities in various types of water systems, e.g., oligotrophic lakes; mountain ponds, alkaline streams, snow melt tributaries, etc.;
- segregation of natural processes such as eutrophication from pollution-enhanced acceleration of the processes;
- sedimentation from chemical reaction of pollutants, e.g., ferric hydroxide precipitation from some acid wastes;
- the adsorptive properties of sediments -- how sediments remove pollutants from the water column, how the pollutant is modified in the benthos by biological or chemical processes;
- the interactions of combined pollutants to enhance or lower the effects on the aquatic ecosystem; and

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- the relative contribution of controllable pollutants (point-source) and uncontrollable (nonpoint sources) pollutants must be considered.*

The EPA must recognize that the major pollution remaining in the nation's waters comes from nonpoint sources. Additional point-source controls are expensive and will do relatively little to improve water quality. Let's start working on the real problem.

The time is right for the Environmental Protection Agency to begin working in a cooperative manner with industry and stop its adversarial role towards industry in order that we can start a realistic solution to pollution control. We are ready, willing and able to assist in water quality criteria development -- we simply are prevented from the opportunity to do so.

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*(EPA recently proposed that ammonia should be added to the list of toxic pollutants, (45 Fed. Reg. 803, Jan. 3, 1980). Yet less than 4.5 percent of the ammonia entering the nation's waters would be covered by increased treatment requirements. Spending billions of dollars to remove an insignificant amount of pollution is simply playing politics to appear to solve a problem which realistically is a nonproblem. This is an expensive exercise in futility.)

1. Thurston, Robert V., et al. eds. 1979. A review of the EPA red book: quality criteria for water. Water Quality Section, American Fisheries Society, Bethesda, Maryland.

2. NRDC v. Train, 8 ERC 2120 (DDC 1976).

3. ________ 1978. Evaluation of statistical methodology used in deriving EPA water quality criteria. Prep. for Am. Petrol. Inst. by Raytheon Company, Portsmouth, Rhode Island.

THE IMPLEMENTATION OF WATER QUALITY CRITERIA AND STANDARDS IN THE DISTRICT OF COLUMBIA

John V. Brink

D.C. Department of Environmental Services
1510 Overlook Ave., S.W.
Washington, D. C. 20032

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THE IMPLEMENTATION OF WATER QUALITY CRITERIA AND STANDARDS IN THE DISTRICT OF COLUMBIA

To carry out the intent of the Federal Water Pollution Control Act Amendments of 1972, and now the Clean Water Act of 1977, the Environmental Protection Agency published regulations and requirements with regards to adopting water quality standards. These regulations are published in the Code of Federal Regulations, Title 40, Part 35, Section 35.1550(b). The regulations are as follows:

35.1550 (b) The water quality standards of the State shall:

(1) Protect the public health or welfare, enhance the quality of water and serve the purpose of the Act;

(2) Specify appropriate water uses to be achieved and protected, taking into consideration the use and value of water for public water supplies, propagation of fish, shellfish and wildlife, recreation

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purposes, and agricultural, industrial and other purposes, and also taking into consideration their use and value for navigation; and

(3) Specify appropriate water quality criteria necessary to support those water uses designated pursuant to 35.1550(b)(2).

The above regulations require the District of Columbia to specify appropriate water quality criteria necessary for propagation of aquatic life and for protection of domestic water supplies. Without adopting such criteria along with our water quality standards the Administrator, EPA, may deny approval and promulgation of these standards as they would not be consistent with the applicable requirements of the Act. Upon the Administrator's notification to include criteria set forth by EPA, if the District refuses to include such, the Administrator of EPA can promulgate standards and criteria pursuant to provisions of the Clean Water Act.

The District's position relative to what criteria are considered to be and what standards are considered to be have been the subject of considerable "in house" discussion in which there is still not 100% agreement. This is not because we cannot define criteria or define standards. Water quality criteria do not have, in our opinion, direct regulatory use but they are to be used as the basis for judgement in water quality considerations in our water pollution control program. A water quality standard on the other hand is established to use as a basis for regulation or enforcement. A standard may differ from a criterion because of prevailing local and natural conditions, economic conditions, or degree of water quality needed for prevailing or desired present or future water use.

Herein lies our problem. Our political leaders and certain of our agency heads are concerned that our EPA counterparts in their administration of federal legislation will enforce national criteria as standards to be complied with by the District of Columbia. Their concerns are not without justification since it appears to us now that that is what EPA is about to do.

Water quality standards should be established based on local stream conditions and the public's expressed desires relative to water use. However, that is not the case in the Washington Metropolitan area. Standards are approved by EPA on a State by State basis with inadequate consideration as to the practicability of the standard as it relates to local conditions or the adverse economic effects on the upstream discharger if required to meet impractical downstream standards established by another State.

For example, within the District of Columbia we have established for the Potomac immediately above the District line a water quality standard of 1000 fecal coliform/100 ml. Immediately below the downstream Maryland-D.C. line the State of Maryland has established a much lower fecal coliform standard for swimming water, and classified the stream as Class I water. Both standards have Region III's approval, however, neither Maryland, Virginia nor D.C. has a program that can assure this class of water entering D.C.

waters nor has EPA to date attempted to require programs directed toward achieving such a standard.

We in the District are now faced with a reproposed standard by Maryland which establishes a 5 mg/liter minimum Dissolved Oxygen criterion for the tidal Potomac immediately below the District line. Maryland appears to have abandoned criteria like those presented in the Blue Book (NAS 1973) in favor of the present 5.0 mg./liter minimum value. The disadvantage of the revised national criteria is that they fail to consider natural seasonal conditions in a given water body. We in the District do not believe that we know what the assimilative capacity of the Potomac at low flow conditions at Washington may actually be. We are still trying to get a handle on the background dissolved oxygen demand of the waters entering the District from point and non-point sources. The basis for establishing the 7 day - 10 year low flow requirement imposed by EPA throughout the country and upon which standards are based has never been explained to our satisfaction.

We are concerned that the District and its environs may be forced to reap negative environmental benefits from regulatory requirements directed toward achieving statutory goals of zero discharge of pollutants and "fishable - swimmable" waters everywhere.

Present policies that are causing considerable expenditures of public funds for projects directed toward avoiding a possible violation of water quality during some future drought period are based on theoretical mathematical models and need to be reexamined. The net environmental benefits from costly projects and the high treatment technologies required for the Washington Metropolitan area needs to be reassessed. Are we actually making progress here in D.C. if we require the utilization of excessive amounts of costly energy and large amounts of chemicals in the production of tremendous amounts of hard-to-dispose-of sludge in order to meet water quality standards based on national criteria not related to D.C. surface waters or desired water uses.

This is an area that requires considerable thought and analysis before the District of Columbia and its environs embark on making any major revisions to present water quality standards. It is for this reason that the District may find itself on a collision course with EPA if proposals to enforce national criteria as standards on D.C. result in excessive project and operating costs without adequate assessment of the environmental benefits to be gained.

Our efforts to date to achieve improved water quality have been successful. Our regional wastewater treatment plant now provides full secondary treatment. All dry weather sewage flows now reach the wastewater treatment plant. Problems associated with pollutional discharges from our combined sewerage system and storm water flows are still with us. Studies are underway to assess the effect of these discharges and to assess the residual pollutional loadings that may be acceptable for discharge to the Potomac River.

The need for advanced waste treatment is being analyzed further in the light of recent information and our laboratory analytical work and stream monitoring activities have been upgraded.

Excellent fishing has returned to the river and our Interstate Commission on the Potomac River is now advocating swimming in the upstream Potomac within the District. Truly, we have come a long way in improving Potomac River water quality. The time has now come to assess our progress and to evaluate costs and environmental trade-offs associated with the land, air and water as they relate to pollution abatement strategies developed to meet future water quality standards.

PROBLEMS IN IMPLEMENTATION OF US EPA WATER QUALITY CRITERIA INTO STATE WATER QUALITY STANDARDS

G. Fred Lee* and R. Anne Jones*

Department of Civil Engineering
Colorado State University
Fort Collins, CO 80523

ABSTRACT

PL 92-500, the 1972 Amendments to the Water Pollution Control Act, requires that the US EPA develop water quality criteria which can serve as a basis for development of state water quality standards. At this time many states are attempting to adopt the numeric values of the US EPA July 1976 Red Book criteria as their water quality standards. This approach is not technically valid and can result in expenditure of unnecessarily large amounts of money in the name of water pollution control. The basic problem with this approach is that the US EPA criteria are generally based on an assumption of the worst case conditions of 100% available forms of contaminants and chronic exposure of organisms to the contaminants. While this approach is administratively simple and generally protective of aquatic systems, it will not result in the best use of the financial and energy resources for water pollution control. This paper discusses some of the problems of trying to implement the US EPA Red Book criteria into state water quality standards.

In order to be legally enforceable, a standard must be analytically measurable. There are several examples of contaminants which cannot be analytically measured at the criteria levels. Standards must also take into account the fact that contaminants exist in aquatic systems in a variety of forms only some of which are available to affect aquatic organisms. Further, since physical and chemical contaminants typically affect beneficial uses of water as a function of not only concentration but also duration of exposure, water quality standard violation should not be

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*Also with EnviroQual Consultants & Laboratories,
Fort Collins, CO 80525

assessed based on a single concentration value or some statistical average value, but rather on the concentration of available forms - duration of exposure coupling. Water quality monitoring programs must be designed to take this into account. Since chemical analytical procedures rarely distinguish between available and unavailable forms of contaminants, increasing use will have to be made of aquatic organism bioassays in assessing the potential for water quality impairment to occur as a result of contaminant discharges.

The failure of the US EPA to provide the kinds of information needed by state and local pollution control agencies to properly implement their water quality criteria into technically valid water quality standards is causing widespread concern and consternation in the water pollution control field about the ability of this agency to properly implement the provisions of PL 92-500.

INTRODUCTION

In accord with the provisions of PL 92-500, the 1972 Amendments to the Federal Water Pollution Control Act, the US EPA must develop water quality criteria which are to represent the current state of knowledge on the impact of physical, chemical, and biological contaminants on water quality. The US EPA has, by necessity, adopted "worst case" criteria (US EPA, 1976 Red Book); that is, these criteria are generally levels of contaminants in 100% available forms which are, based on current knowledge, safe for sensitive organisms to be exposed to for their entire lifetimes or during critical stages of their lives. While in the Foreword to the US EPA Red Book R. Train, then US EPA Administrator, stated:

> "Guidelines to implement the consideration of criteria present in this volume in the development of water quality standards, and in other water-related programs of this Agency, are being developed and will be available for use by the states, other agencies, and interested parties in the near future."

to date, some four years after the Red Book was released, the US EPA has still not provided appropriate guidance in this regard. According to current US EPA policy, state water quality standards for the total concentration of a contaminant in a water must be at least as stringent as the worst case US EPA water quality criteria or the state must demonstrate why a more lenient standard should be accepted. Many states, which are currently in the process of developing or revising their water quality standards, are adopting the numeric values of the US EPA July 1976 criteria as their water quality standards.

PL 92-500 also set as a national goal zero pollutant discharge. While not specified in the Act, the US EPA water quality criteria will certainly serve as a basis for distinguishing between discharge of a "pollutant" and discharge of a "contaminant." A "contaminant" is any material added to a water. A "contaminant" becomes a "pollutant" when its critical ("no impact") concentration - duration of exposure coupling is exceeded, causing an adverse impact.

When PL 92-500 was passed, there were many who questioned the economic and technical feasibility of achieving zero pollutant discharge and using numeric values of the worst case water quality criteria as state water quality standards. However, as late as 1977 when PL 92-500 was amended, the mood of the U.S. and, most importantly, its national, elected representatives, has been such that the basic philosophy of the Act remained essentially unchanged. Today, as the date for achieving zero pollutant discharge draws near, there are many more who question not only the desirability of achieving this goal because of its economic and social ramifications but also, more importantly, the desirability of adopting the US EPA water quality criteria of July 1976 as state water quality standards to in turn serve as a basis for waste load allocations in water quality limited waterbodies.

The General Accounting Office (GAO) has recently reviewed, in a report entitled "Many Water Quality Standard Violations May Not Be Significant Enough to Justify Costly Preventive Actions" (GAO, 1980), the potential cost-effectiveness of advanced wastewater treatment (AWT) as a means of further reducing contaminant concentrations in domestic wastewater. It has recommended, according to a report entitled, "GAO Questions $10 billion cost for Advanced Wastewater Treatment" (Air/Water Pollution Report, 1980), that

> "Congress should declare a moratorium on advanced wastewater treatment (AWT) projects by withholding funding for treatment beyond secondary until the Environmental Protection Agency 'can clearly show what ecological, social, and public health benefits are being realized by the various levels' of AWT . . ."

PROBLEM IN USING WORST CASE CRITERIA AS WATER QUALITY STANDARDS

The basic problem with using the numeric values of the US EPA July 1976 water quality criteria as the legally enforceable water quality standards is that, for the most part, natural water systems do not represent worst case situations. As discussed by Lee et al. (1980a), there are a number of characteristics of chemicals as well as natural waters which reduce the potential impact of a chemical on water quality (fisheries, drinking water supply, recreational utility, aesthetic quality, etc.)

CONTAMINANT AVAILABILITY

Many chemical contaminants exist in aquatic systems in a wide variety of forms, only some of which may be available to cause an adverse effect on beneficial uses of the water. While the US EPA water quality criteria generally try to take into account the impacts on toxicity of such parameters as hardness and alkalinity, they disregard the more significant impacts of a variety of chemical reactions on toxicity of chemical contaminants. One of the most important reactions which renders many potentially hazardous chemicals unavailable (non-toxic) or less available to aquatic life is sorption. (It should be noted that sorption is frequently mislabeled as "complexation" by various investigators in explaining the change in behavior of a contaminant, such as a heavy metal, when placed in natural water systems; this change is, however, more likely due to sorption on colloidal materials.) Several years ago, the authors conducted a major study on the significance of dredged sediment-associated contaminants in affecting water quality during dredging and disposal operations (Jones and Lee, 1978; Lee et al. 1978; Jones et al. 1980). They found, as expected, that even though sediments often contain very high concentrations of a wide variety of chemical contaminants, they are not typically released in significant quantities to overlying waters. Even when dredged sediment are disposed of in open water, rarely would sufficient amounts of contaminants be released to cause mortality to pelagic organisms of the disposal area.

It is not only sorption on sediments which renders contaminants unavailable or less available, but also sorption on other materials as well. As discussed by Lee (1975), iron and aluminum hydrous oxides play extremely important roles in controlling the availability of many chemical contaminants in aquatic systems. Ferric hydroxide tends to form amorphous colloidal particles (floc) which effectively sorb many contaminants of concern. Contaminants sorbed on such material have markedly different bioavailabilities than contaminant forms conventionally used in laboratory toxicity and bioavailability testing.

Complexation also plays a role in affecting the toxicity of some contaminants. Soluble forms of contaminants, especially heavy metals, can be complexed, rendering them unavailable or less available to affect water quality. Complexation, however, as discussed by Sanchez and Lee (1973) can also act to make certain contaminants more available by causing their desorption from particulate matter and creating soluble complexes which are more available. Other potentially important reactions affecting contaminant availability include oxidation-reduction, photolysis, hydrolysis, precipitation, acid-base, gas transfer, and biotransformation. The significance of the thermodynamics and kinetics of these reactions in affecting contaminant availability have been discussed by Lee et al. (1980a).

An example of the changes in availability of contaminants as a function of characteristics of the water, was provided by the work of Morrison and Post at Colorado State University (Morrison et al. 1979). They conducted a series of laboratory bioassays using waters from various reaches of the Poudre River (Colorado) spiked with copper, cadmium, silver, and ammonia to determine 96 hour LC_{50} values for trout and fathead minnows. The almost pristine water of the Poudre River flows from the Colorado Rockies into the plains of the Front Range of Colorado. During much of the year, within a short distance of the mountains, the water from the Poudre is diverted for irrigational purposes to the point that at some times of the year, this diversion depletes the river of water. Except during flood-flow periods, by the time the water passes through Fort Collins, a distance of about 20 river miles, the pristine water has essentially been completely replaced by domestic wastewaters which have been discharged to the river, irrigation return flows, and shallow groundwater seepage arising from irrigation. This markedly changes the chemical characteristics of this water. Morrison et al. (1979) found that the 96 hour LC_{50} values of the chemicals tested in samples of Poudre River water as it leaves the mountains in a pristine state, closely match those reported in the Red Book for many of the contaminants tested and the organisms used. The toxicities of the same chemicals to the same types of organisms in water taken from the Poudre River downstream of the City of Fort Collins were an order of magnitude less. While the reason for this decrease in toxicity is not clear, a substantial part of the difference appears to be due to differences in alkalinity and hardness of the river water. It is possible that it also relates to the increased presence of organic compounds and other particulate matter which tend to sorb and complex heavy metals.

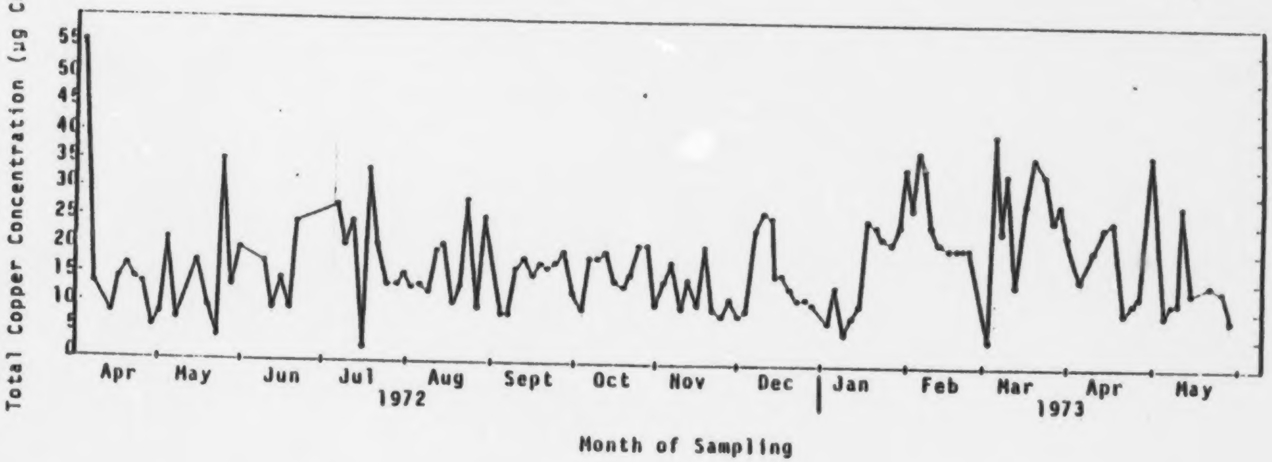
While, for many waters, measurement of the "soluble" fraction of a contaminant generally gives a better estimate of the available fraction than measurement of the total concentration, it can result in either overestimation or underestimation of the potential toxicity or bioaccumulation of a contaminant. For example, as discussed previously, ferric hydroxide colloidal material is an efficient scavenger of many potentially hazardous contaminants. However, some of these particles are small enough to pass through the standard 0.45 μ pore size filter used for separation of "soluble" and "insoluble" components, causing some of the sorbed colloidal material to be measured as soluble. At this time there is no simple, chemical test that can be used to distinguish between available and unavailable forms of contaminants in aquatic systems. As discussed by Lee et al. (1980a,b), the only way to properly distinguish between these groups is through the use of aquatic organism bioassay test procedures.

VARIABLE CONCENTRATIONS

The second reason that the US EPA water quality criteria are difficult to use directly as water quality standards is the fact that these criteria, because they represent safe concentrations under worst case conditions, assume that aquatic organisms receive chronic exposure to the level of contaminant found at a particular sampling time. Usually chronic exposure means lifetime exposure or exposure for significant parts of important life stages of aquatic organisms. As discussed by Lee _et al_. (1980a), it is rare that the concentrations of contaminants associated with any discharge are constant. Typically these concentrations are continuously varying by orders of magnitude over a 24-hour cycle. Further, the waters into which they are discharged, particularly streams and rivers, have varying flows depending on season, time of day, degree of regulation, etc. The net result is that the concentrations of many contaminants of greatest concern are highly variable in many aquatic systems. Lee and Jones (1979b) conducted an evaluation of the water quality-related data collected on Lake Ontario and its U.S. tributaries during the International Field Year for the Great Lakes (IFYGL). They found that water samples collected every three days from the mouth of the Genesee River over a one-year period had concentrations of total copper which varied from about 2 to 55 µg/l, as shown in Figure 1. Since available copper is typically toxic to some aquatic organisms in this range, if the standard is established in the 15 to 25 µg/l range, there would be frequent excursions above the standard for short periods of time. As discussed in a subsequent section, this may or may not have an adverse impact on aquatic organisms.

In a study recently completed by Lee _et al_.(1980c) in which chlorine concentrations were measured every several hours over a four-day period on water samples collected from the Arkansas River at the Pueblo, CO, sewage treatment plant discharge, it was found that chlorine concentrations ranged from 0.02 to 0.26 mg/l Cl_2. During the same study period, the daily water temperature fluctuated by 10 C °. Based on the experience of the authors, these kinds of variations are not uncommon in aquatic systems. Studies conducted by Kodak of Colorado on the toxicity of ammonia to adult fathead minnows (Morrison _et al_. 1979) has shown that these organisms could be exposed to 0.1 mg/l un-ionized ammonia for a period of 28 days without experiencing any deaths. Further, when they were exposed to ammonia concentrations of 0.3 mg/l un-ionized ammonia for several days in sequence and then to the 0.1 mg/l un-ionized ammonia base level, there were no deaths of organisms in the 28-day test period. Exposure of fathead minnows to 0.3 mg/l un-ionized ammonia for on the order of one week will result in organism mortality. Exposure to 0.4 mg/l un-ionized ammonia will result in fathead minnow

Figure 1. Total Copper Concentrations at the Genesee River Mouth (Lake Ontario)



After Lee and Jones (1979b).

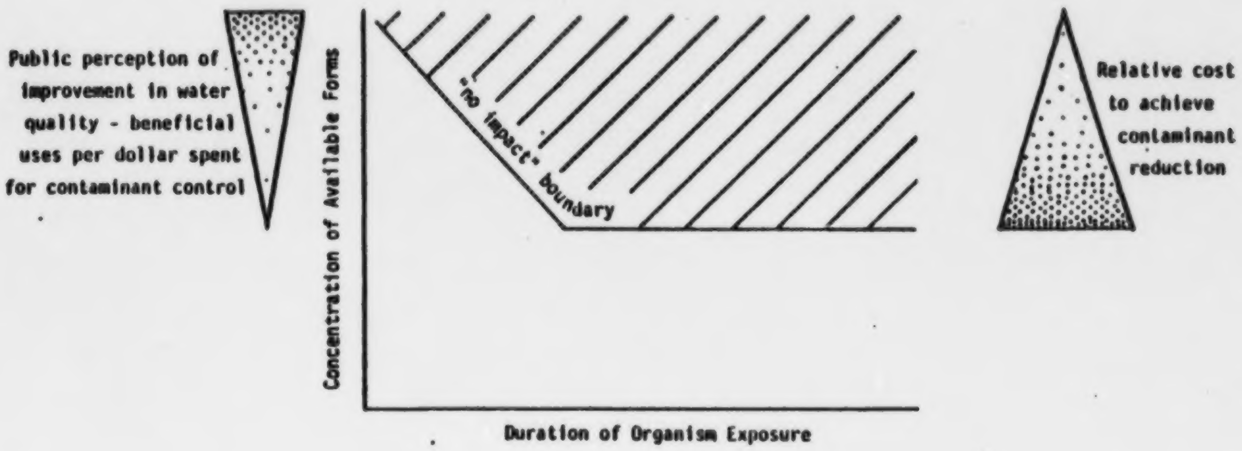
mortality in several days. (It should be noted that the endpoint in these bioassays was death; there may be some chronic impacts on adult fathead minnows or other adverse effects on other life stages of fathead minnows at these levels.)

Under conditions where the concentrations of a contaminant downstream of a point source are typically at or near the Red Book criterion value, there could be excursions of concentrations to levels an order of magnitude greater than the Red Book value. If, as is the typical case in many states, exceeding the standard in a single sample constitutes a water quality standard violation, it is possible that a water quality standard is exceeded without there being any adverse impacts on water quality. To properly implement water quality criteria into water quality standards, consideration must be given to the fact that concentrations of contaminants in available forms can greatly exceed, for some period of time, the criterion value without adverse effect on the beneficial uses of the water. These situations create considerable misunderstanding on the part of many water pollution control practitioners, governmental agencies, industrial representatives, and others. It is difficult for persons with limited knowledge of aquatic chemistry and toxicology to understand that there are times when the concentrations of certain contaminants in rivers, streams, lakes, or estuaries are considerably above the Red Book, worst case criteria value without there being a significant hazard to aquatic life. In situations such as these, it is not unusual for electroshocking or other fish population evaluation techniques to show that there are appreciable concentrations of organisms present in the region with the apparently excessive concentrations of chemical contaminants. This should result in public opposition to attempts to impose Red Book criteria as single, fixed numeric standards which, if exceeded in any sample of water, signals a violation of the state pollution control regulation.

CONCENTRATION - DURATION OF EXPOSURE RELATIONSHIPS

The fact that contaminant concentrations in a given aquatic system are often highly variable and can fluctuate considerably above and below chronic safe levels necessitates the consideration not only of concentrations of available forms of contaminants relative to chronic safe levels in assessing hazard and hence in developing standards, but also of the length of time that organisms are exposed to potentially hazardous (i.e., greater than chronic safe) levels. As discussed by the authors (Lee et al. 1980a,b) it is well known that organisms can be exposed to concentrations of available forms of contaminants substantially higher than the US EPA Red Book criteria without there being an adverse effect on beneficial uses of the water. Figure 2 illustrates this relationship showing that for shorter durations of organism exposure, "no impact" concentrations are higher than those at the chronic exposure plateau.

Figure 2. Relationships among Potential Impact of Contaminants on Water Quality - Beneficial Uses of the Water, Relative Costs to Achieve Various Degrees of Water Quality Improvement, and Perceptibility of Various Degrees of Improvement in Water Quality to the Public.


Public perception of improvement in water quality - beneficial uses per dollar spent for contaminant control
Concentration of Available Forms
"no impact" boundary
Duration of Organism Exposure
Relative cost to achieve contaminant reduction

An example of the importance of considering the duration of exposure in assessing impact of contaminants is provided by chlorine. The chloramines that are present in domestic wastewaters as a result of chlorination for disinfection purposes will, at concentrations of approximately 0.5 mg/l, kill certain types of fish within a few hours. The 96 hour LC_{50} for the chloramines typically present in domestic wastewater effluent after chlorination is on the order of 0.1 mg/l, while the chronic - lifetime exposure safe concentration is on the order of 0.005 mg/l. There are two orders of magnitude difference between the level which will cause acute lethality in aquatic organisms in a few hours and the chronic safe concentration. A similar situation exists for ammonia. The chronic safe concentration for warm water fish is generally recognized to be on the order of 0.05 mg/l un-ionized ammonia, while fish of the same type can spend months in waters having concentrations on the order of 0.1 to 0.2 mg/l un-ionized ammonia.

As discussed in the American Fisheries Society critique of the Red Book ammonia criterion (Willingham et al. 1979), ammonia and many other contaminants at concentrations between the acute lethal concentration (i.e., that which will kill the organisms in four days) and the chronic safe value (i.e., Red Book value), affect organism characteristics such as weight, growth rate, susceptibility to disease, etc. While it is generally believed that the majority of the public would strongly support the kind of control programs which would prevent the death of fish, it still remains to be seen whether or not the public in general wishes to pay the enormous costs, either through their taxes or increased price of goods, for the control of contaminants to the point of eliminating the potential of chronic toxicity manifested in decreased rates of growth, slightly less healthy fish, a somewhat impaired reproduction rate, etc. While there may be considerable sympathy for control of contaminants to prevent acute lethal toxicity, it is highly doubtful that the public as a whole will pay for contaminant control programs to provide the ultimate, based on amount of suitable habitat, in fisheries for a particular waterbody. It is important that the public be informed as to the potential benefits that will be derived from controlling contaminants to the degree necessary to achieve Red Book criteria.

AVOIDANCE

Another factor that causes considerable confusion in assessing the actual significance of a particular point source discharge of a contaminant as a cause of impairment of beneficial uses of water, and that renders inappropriate the blanket comparison of a concentration of a contaminant in a water with worst case numeric standards, is the avoidance reaction of fish and other mobile aquatic organisms. For example, in the case of chlorine, it is well known that fish will avoid high concentrations of chlorine, given the opportunity to do so. While avoidance is well recognized as a phenomenon by which fish can naturally minimize their exposure to adverse concentrations of some chemicals, there is also an attraction of fish to some point source discharges, for example, some with elevated temperatures. This could cause an increased exposure to some contaminants over that which would normally occur. There is insufficient information available at this time to know whether during winter conditions a chlorinated domestic wastewater effluent would, because of its elevated temperatures, attract the fish more, or because of high chlorine concentrations, repel fish. The information base is grossly inadequate to make any conclusions as to whether avoidance or attraction to a particular effluent will be the dominant behavioral pattern of a certain type of fish in a certain location.

ACCLIMATIZATION

Certain aquatic organisms can become acclimatized to elevated levels of available forms of certain contaminants if they have been exposed to them previously or continuously. Such organisms can tolerate without any apparent, significant harm, relatively higher concentrations of the contaminant than can the same kind of organisms suddenly introduced into the higher concentration waters. This is another factor that should be considered in assessing the hazard associated with a contaminant as it is one which tends to reduce the potential for impairment of beneficial uses of a water by a chemical contaminant.

ANALYTICAL CONSIDERATIONS

In order for water quality standards to be enforceable, it must be possible to collect an appropriate water sample and to reliably measure the contaminant in a water sample at and below the level of the standard. Some of the US EPA Red Book criteria are below reliable analytical detection limits. For example, sulfide is toxic to fish eggs at a few µg/l. Not only is it extremely difficult to collect a proper water sample from the sediment/water interface where many types of fish eggs would be found, but also it is difficult to reliably measure sulfide in this water at these levels. Similarly, while PCBs can be measured down to 2 to 3 ng/l in water under optimum conditions, the PCB criterion is 1 ng/l. Since for PCBs, the concern is their concentrations in edible fish flesh, it has been recommended by the American Fisheries Society PCB criterion review panel (Veith et al., 1979) that PCB standards be for fish flesh content rather than for water. This can be readily accomplished because the Food & Drug Administration has PCB limits for edible fish flesh and PCBs can be readily measured at and below the FDA limits. The same approach should be used for other contaminants for which the concern is bioconcentration-bioaccumulation. Lee and Jones (1979a) discuss considerations that must be made in interpreting chemical water quality data.

As indicated previously, the US EPA-Washington has been delinquent in providing appropriate guidance for implementation of the Red Book criteria into state water quality standards. If it is the intent of the US EPA-Washington to have the states use the numeric, worst case Red Book criteria values directly as water quality standards, then it has been grossly negligent in insuring that suitable analytical techniques are available for measuring contaminants at and below criteria levels. In the early 1970s it was called to the attention of the US EPA-Washington by Lee et al. (1974) and others that a number of the then proposed criteria would be essentially worthless as standards unless substantial funding were made available for development of suitable analytical procedures. Since that time, the US EPA has devoted inadequate attention (funding) to this area with the result that states are having great difficulties in developing appropriate, enforceable standards. Those responsible for criteria promulgation have taken the attitude that criteria promulgation should not be dependent upon the availability of analytical procedures, and frequently express the view that analytical procedure development is the responsibility of other groups within the US EPA. It is clear that if the Red Book criteria are to be used as state water quality standards, a substantial effort should be undertaken in the immediate future to develop the necessary analytical procedures.

APPROACH FOR DEVELOPMENT OF WATER QUALITY STANDARDS

As discussed in the Introduction, PL 92-500 and its 1977 amendment set forth a number of target dates for attainment of various degrees of pollution and contaminant control in US waters. The most immediate goal is that of attaining swimmable, fishable waters in the US by 1983. In this regard, there seems to be some confusion as to what is meant by "fishable" water. Environmental activist groups, the federal and some regional EPA groups, as well as some state pollution control agencies, have taken the attitude that "fishable" means the ultimate in fisheries potential. There is even question about what the "ultimate" in fisheries means; it could mean large populations of desirable game fish or a more "balanced" fishery containing a certain diversity of species containing both desirable and less desirable fish. In either event, it does imply that the numbers and types of fish present in any reach of the river are controlled exclusively by the habitat limitations.

Pollution control efforts already undertaken in the US have essentially eliminated the grossest of pollution in the Nation's waters, such that there are no longer the widespread problems of fish kills due to discharges of contaminants. As indicated earlier, the emphasis in pollution control is shifting toward attainment of the ultimate in fisheries (zero pollutant discharge); eliminating the subtle effects of polluction, such as impaired growth rate, egg production, altered swimming behavior, etc.

As major efforts in this direction begin, as they are with the classification of streams and other waterbodies, the public will begin to realize that the improvements in beneficial uses of waterbodies obtained per dollar spent in pollution control will be greatly diminished. Figure 2 illustrates this concept as it shows substantial increases in cost for achieving lower and lower concentrations (to the chronic safe levels) of available forms of contaminants. At the same time, the impacts of the further reduction of contaminant concentrations on water quality - beneficial uses, become less. Because of the factors discussed previously (such as, reactivity, duration of exposure, etc.) however, it is not necessary in all cases to reduce contaminant concentrations to chronic safe levels in order to provide protection of beneficial uses of the water of concern. It is crucial with the current inflation and energy problems that funds available for pollution control be spent in such a way as to gain the most benefit per dollar spent. This will necessitate development of water quality standards which take into account environmental chemistry and physics of contaminants in natural waters, and the impact of contaminants on water quality - aquatic organisms.

HAZARD ASSESSMENT APPROACH FOR WATER QUALITY STANDARDS DEVELOPMENT

Lee et al. (1980b) have presented a discussion of the use of a "hazard assessment" approach for developing technically valid, cost-effective, and environmentally protective water quality standards. This approach employs the US EPA Red Book criteria to flag waters having potential water quality problems. If a water contains concentrations of total contaminants above these chronic exposure safe levels, the source(s) of the contaminants must then conduct a hazard assessment to determine if the apparently elevated concentrations impair beneficial uses of the water. In a selective, sequential testing scheme, such as is described by Lee et al. (1980a), the environmental chemistry-fate, including availability, and toxicology - organism response, and behavior of the contaminants in the system of concern are evaluated. If it is determined that the levels of contaminants present are or could be impairing beneficial uses of the water, then the source(s) must reduce their inputs sufficiently so as to reduce the hazard to an acceptable level.

Some advocate use of a statistical approach for establishing water quality standards. For example, the State of Colorado uses the mean concentration plus one standard deviation as an indication of a violation of a water quality standard for certain parameters in Colorado waters. This, as well as other simple statistical approaches, is not technically valid for use to assess violations of water quality standards or adverse impacts of contaminants on beneficial uses of waters. It is known that aquatic organisms are not concerned with the mean concentrations or, for that matter, with the standard deviation. As discussed above, the impact that a contaminant has on beneficial uses of a water depends on the amount of available forms of the contaminant and the duration of organism exposure.

This must be the basis by which technically valid water quality standards are developed. Further, water quality monitoring programs should be conducted to generate data of this type. The typical sporadic or regular but infrequent sampling will not generate the information needed to estimate potential impact. It is far better to conduct water quality monitoring programs for point source discharges with short-term, high-frequency sampling during periods exhibiting worst case conditions of the receiving waters, i.e., low flow, high temperature (or low temperature) than periodic sampling at weekly, monthly, or other infrequent intervals.

HIERARCHY OF WATER POLLUTION CONTROL

One issue with which water pollution managers have had to deal for many years is the dissolved oxygen concentrations in receiving waters downstream from wastewater treatment plants. Table 1 presents the general hierarchy of treatment that is traditionally followed for the control of contaminants in domestic wastewaters. At level A the focal point is directed toward floatable and settleable solids. The control program is usually accomplished through primary treatment involving sedimentation and skimming of floating materials in a settling tank. Appreciable amounts of various types of contaminants which can have adverse effects other than impairing aesthetic quality, are also removed through primary settling although primary emphasis of this treatment is to control materials which would have caused objectionable accumulations of contaminants in the receiving water sediments and most importantly,materials that would float on the surface and be readily visible. With this level of treatment, a fishery will not likely be maintained within close proximity downstream of the effluent discharge unless the flow of the effluent is sufficiently small compared to the flow in the receiving water.

At level B, or secondary treatment, the emphasis in treating domestic wastewaters is traditionally directed toward control of oxygen depletion in the receiving waters and is accomplished by activated sludge or trickling filter biological treatment. The degree of dissolved oxygen depletion in receiving waters is a function of the degree of BOD (biochemical oxygen demand) removal and the relative flows of the receiving water and the wastewater. By tradition there are three levels of treatment that are usually considered in connection with BOD control. The first of these is the elimination of noxious odors associated with hydrogen sulfide, i.e., achieving from 0 to several mg/l DO in the receiving waters. No attempt is made at this level of BOD control to maintain a fishery below the effluent discharge, although some rough fish, such as carp and suckers, are usually

TABLE 1. WATER QUALITY MANAGEMENT FOR CONVENTIONAL DOMESTIC WASTEWATERS

Level	Problem	Control Method
A	Floating and Settleable Solids	Primary Settling
B	Oxygen demand	Secondary Treatment (Activated Sludge - Trickling Filter)
B_1	Sulfide odor	Trace - 4 mg/l DO
B_2	Maintain good fishery	4 mg/l DO minimum
B_3	Maintain ultimate fishery	Min. DO 5 mg/l (or 80% saturation) $NH_3 \lesssim 0.05$ mg N/l NO_2^- ?

able to survive under these conditions.

For many years dissolved oxygen (DO) minimum levels on the order of 4 mg/l have been accepted as the level that should be present to maintain a warm water fishery. To maintain this dissolved oxygen level could be considered the second or intermediate level of BOD control. While the ultimate in fisheries may not be maintained in the vicinity of the discharge with this level of treatment, it is generally suitable for a good fishery within habitat constraints.

The third level represents the ultimate in BOD control. There are two basic approaches that are used. One is the maintenance of the US EPA Red Book minimum of 5 mg/l DO. The National Academies' committees concerned with dissolved oxygen concluded that any time the dissolved oxygen concentration was less than about 80% of saturation, the fish were being exposed to adverse conditions. While they are not killed at less than 80% of saturation unless the DO gets to a critical level, they are more susceptible to disease, tend to grow at slower rates, etc. Considerable controversy is developing today concerning the cost-effectiveness of maintaining the US EPA minimum 5 mg/l DO versus the more traditionally accepted 4 mg/l DO. This controversy is exemplified by the situation that has developed between the State of Ohio and the US EPA, where this State proposed a minimum water quality standard of 4 mg/l DO. The US EPA insisted that it conform to the Red Book number of a minimum value of 5 mg/l. The State of Ohio pollution control agencies filed suit against the US EPA for what they felt was an inappropriate decision which would lead to unnecessary expenditure of Ohio residents' funds. This situation is an example of the controversy that is going to be developing with respect to many of the US EPA water quality criteria. As discussed elsewhere, it is important to know what improvement in beneficial uses will be attained with increasingly stringent standards and whether the difference is of sufficient value to spend the money necessary to attain the standards or if the available funds would be spent more wisely in other areas.

NEED FOR RESEARCH

It was recognized by many in the early 1970s with the development of the water quality criteria provision of PL 92-500, that in order to bring about a cost-effective conversion of these criteria into state water quality standards, there would have to be a substantial research effort undertaken to provide the kinds of information necessary to avoid the kinds of implementation problems that exist today. Unfortunately, essentially concomitant with the adoption of PL 92-500, there was a significant curtailment in the research effort by the US EPA and other governmental agencies in two of the areas that were crucial to implementation of the US EPA criteria into state water quality standards, namely, aquatic toxicology and aquatic chemistry. The development of information in these two areas during the past half-dozen or so years has been far less than that necessary to begin to implement Red Book criteria into standards.

Another serious deficiency in our current state of knowledge relative to implementation of Red Book criteria into water quality standards is in the understanding of fish behavior. Until we gain a much better understanding of the factors that influence the presence or absence of fish in a particular region of a lake, river, stream, or estuary, we are going to continue to find a significant number of apparent anomalies of finding fish populations in regions which have apparently excessive concentrations of chemical contaminants which should be adverse to them. As discussed previously, there is also need for work in analytical procedures development for those chemicals which cannot currently be measured at or below criteria levels.

The US EPA management, in the opinions of many, has and continues to tremendously shortchange the public in leading them to believe that there is sufficient information available today to develop meaningful standards from the Red Book criteria.

During the past few years the public funding situation has worsened. The US EPA has focused almost all of its limited research funding for work in this area, on obtaining relatively small amounts of information (compared to the need for information) on the chemicals on the "List of 65" hazardous chemicals developed through the Consent Decree (US EPA et al., 1976). It is the opinion of the authors that the Consent Decree has done significant harm to the development of water pollution control programs in this country. It has diverted valuable resources away from the areas that need to be researched to implement contaminant control programs for the classical, known contaminants, such as, ammonia, chlorine, etc., and has forced the focus toward work on a group of exotic, rarely encountered contaminants.

CREDIBILITY OF GOVERNMENTAL AGENCIES IN THE WATER POLLUTION CONTROL FIELD

Local public pollution control agency staff and elected public officials in general have a relatively poor opinion of the ability of state and federal governmental agencies to develop technically valid, cost-effective approaches for achieving certain degrees of designated water quality in the waters of concern to a community. An example of this is the aforementioned report by the General Accounting Office (GAO) questioning the cost-effectiveness of Advanced Wastewater Treatment for domestic wastewater (GAO, 1980).

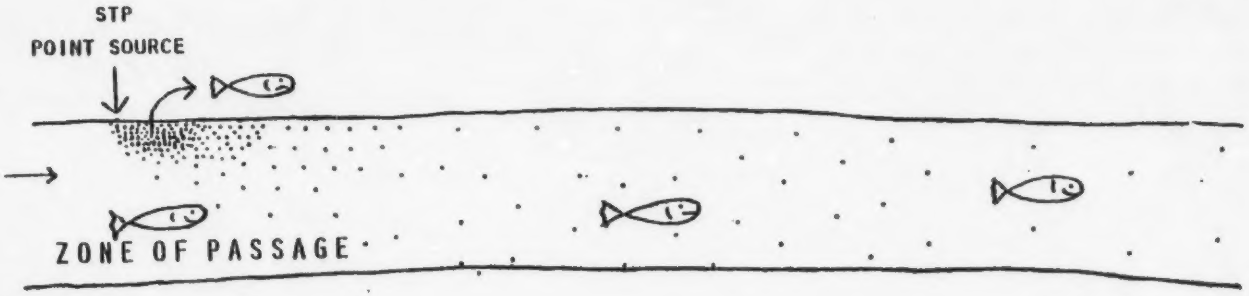
There is a general anti-big government attitude that exists among the public in general in all aspects of life, and is strongly reinforced in the water pollution control field by the US EPA-Washington, various US EPA regions, and state pollution control agencies which try to enforce the adoption of single, fixed numeric values for water quality standards upon which waste load allocations are developed. It certainly does not take much to convince the lay and semi-technically trained individuals that "big brother" in Washington and in the State House in some states is not utilizing the financial resources of the community at the local level to the best possible advantage, when they can readily recognize that the US EPA, through its water quality criteria, has assumed a worst case situation approach and that the concentrations of contaminants can in some instances be considerably above the US EPA Red Book criteria-based standard without significantly adversely affecting the beneficial uses of a particular water.

It is believed that it is fair to state that the US EPA water quality criteria and standards branch in Washington has low credibility among most state pollution control agencies. There are some, possibly most, members of state pollution control agencies (as well as local pollution control agencies) who look on the US EPA as not meeting its obligations in developing meaningful criteria that they can use. The US EPA-Washington has certainly been grossly negligent in this area in failing to provide guidance on how states should handle the numerous problems that are discussed in this paper, as well as others, in trying to implement worst case criteria into water quality standards. There is no question about the fact that this situation is leading to a large number of lawsuits against state pollution control agencies and the US EPA for failing to provide the kinds of information needed to handle these problems. Until the US EPA develops technically valid approaches for handling the various problems that are discussed in this paper associated with using Red Book criteria as state water quality standards, they are going to have little or no credibility in the water pollution control field and they are going to find that decisions on pollution control regulations in this country will be developed in the courts rather than by the technical community.

SUMMARY

In accord with PL 92-500, the US EPA has promulgated water quality criteria which are to be used as a basis for the development of state water quality standards. Many states are attempting to use these criteria values as their water quality standards. The fact that the criteria represent chronic exposure safe concentrations makes it inappropriate in most cases to do this. Figure 3 summarizes a fairly common situation that exists at point source discharges to a stream illustrating this point. Associated with most discharges is an area in which the effluent becomes completely mixed with receiving waters, labeled in Figure 3 as the Zone of Mixing. Contaminant concentrations here

Figure 3. Schematic Representation of Potential Impact of Non-Persistent Chemicals Discharged in Domestic Wastewater Treatment Plant Effluent



ZONE OF MIXING	EXTENT OF POTENTIAL CHRONIC IMPACT	CHRONIC SAFE CONCENTRATIONS
HIGH CONCENTRATIONS, SHORT DURATION OF EXPOSURE, POTENTIAL ACUTE LETHAL TOXICITY, AVOIDANCE	BASED ON US EPA RED BOOK CRITERIA (CHRONIC SAFE LEVELS) DURATION OF EXPOSURE ACCLIMATIZATION	BASED ON US EPA RED BOOK CRITERIA (CHRONIC SAFE LEVELS)

FISHERIES BENEFICIAL USE OF REACH OF RIVER CONTROLLED BY:

HABITAT CHARACTERISTICS POSSIBLE ACUTE LETHAL TOXICITY OF CONTAMINANT	HABITAT CHARACTERISTICS, POTENTIAL CHRONIC TOXICITY OR OTHER IMPAIRMENT DUE TO CONTAMINANT	HABITAT CHARACTERISTICS

are higher than in surrounding waters and for some discharges can cause acute lethal toxicity if organisms spent sufficient time in the area. Concentrations in the Zone of Mixing are decreased by dilution and, to some extent depending on the chemical, by chemical reactions such as those discussed previously. Not infrequently, the shape of the discharge plume will create a Zone of Passage in which fish and other mobile organisms can avoid being exposed to the higher concentrations in the Zone of Mixing; as discussed previously, many organisms tend to avoid, if possible, elevated concentrations of certain contaminants. Even though concentrations in the Zone of Mixing can be acutely toxic, typical pelagic organism exposure would be expected to be short, minimizing potential adverse effect of the contaminant at higher concentrations. The fisheries-related beneficial uses of this portion of the reach of the river would be limited by potential acute lethal toxicity and/or habitat characteristics.

After dilution and reaction have reduced the contaminant concentrations below the acute lethal level, there is a stretch of river in which there is a potential for chronic toxicity based on chronic safe levels (Red Book criteria). It would be expected that this reach would be larger than the Zone of Mixing, extending until reactions of the contaminant and potential further dilution arising from additional water inputs caused its concentration to be decreased to chronic safe levels. Organism exposure here could be longer than in the Zone of Mixing, but the impact would be controlled by the concentration of available forms - duration of exposure coupling. It is also possible that fish could become acclimated to some degree and hence may be able to tolerate the potentially chronically toxic concentrations without adverse effects. Fisheries-related beneficial uses of this portion of the river would be limited by potential chronic toxicity as well as habitat characteristics.

The third zone begins where contaminant concentrations have been decreased to chronic safe levels based on Red Book criteria. Here, only the habitat characteristics limit fisheries - related beneficial uses of the water.

Not illustrated in Figure 3, but nonetheless an important consideration is the fact that the sizes - shapes of these zones continually change depending on flow, discharge rate and concentration, etc., and at times other factors, such as temperature. The concentration at any one point in the discharge plume can show large variations over short periods of time.

Given the above considerations and the fact that for some chemicals, concentrations cannot be reliably measured at the Red Book criteria levels, it is inappropriate to use a chronic safe concentration as a water quality standard, especially if exceeding the standard in any sample collected constitutes a standards violation. Instead, if there is question about the potential impact of a discharge on water quality, a hazard assessment approach should be used to estimate these characteristics of the particular system to determine if the hazard is acceptable in light of the beneficial uses and other limiting factors (such as habitat characteristics, public accessibility, etc.). It is only by using such a site specific approach, considering the characteristics of the chemical, receiving waters, and beneficial uses, that technically-valid, cost-effective, yet environmentally protective standards and contaminant control programs can be developed.

CONCLUSIONS AND RECOMMENDATIONS

The approach for implementation of the provisions of PL 92-500 for the 1980s and beyond is based on state water quality standards. By law, these standards must be based on the US EPA water quality criteria. Many states are using the numeric criteria values directly as water quality standards applied to the total concentration of the contaminant. There are numerous problems with this approach. While the US EPA has developed a variety of water quality criteria and will soon release a new set covering the Consent Decree List of 65 chemicals, the existing criteria (Red Book) were generally developed under or extrapolated to worst case conditions of the contaminant being in 100% available forms, and the organisms receiving chronic exposure; the criteria generally represent chronic exposure safe levels. However, many contaminants exist in natural water systems in a variety of forms such that usually only part of the total concentration is available to affect water quality. Concentrations of contaminants in many systems are continually varying as a result of variable rates of input, flow, dilution, and chemical reaction. Since organisms are affected by contaminants according to concentration of available forms - duration of exposure relationships, it can frequently occur that concentrations well above the chronic safe concentration level without there being an impairment of the beneficial uses of the water. The development and application of standards to provide technically valid, cost-effective, yet environmentally protective contaminant control programs, should be based on the concentration of available forms - duration of exposure relationships for each site of concern.

It is recommended that a hazard assessment approach be used to evaluate alleged violations of water quality standards to determine the water quality significance of apparently excessive concentrations of contaminants.

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HUMAN HEALTH EFFECTS IN DEVELOPMENT OF CRITERIA AND STANDARDS

SECTION III

DEVELOPMENT OF WATER QUALITY CRITERIA

Dr. Frank E. Gostomski

U.S. Environmental Protection Agency
401 M Street, N.W.
Washington, D. C. 20460

ABSTRACT

Section 304(a)(1) of the Clean Water Act requires EPA to publish and periodically revise criteria for water quality accurately reflecting the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, esthetics, and recreation which may be expected from the presence of pollutants in any body of water, including groundwater.

These criteria are also being issued in satisfaction of the Settlement Agreement in Natural Resources Defense Council, et al. v. Train, 8 E.R.C. 2120 (1976), modified, 12 E.R.C. 1833 (D.D.C. 1979). Pursuant to paragraph 11 of that agreement, EPA is required to publish criteria documents for the same 65 pollutants which Congress, in the 1977 amendments to the the Act, designated as toxic under section 307(a)(1). These documents are to contain recommended maximum permissible concentrations for pollutants which are consistent with the protection of aquatic organisms, human health and recreational activities. These criteria documents are scheduled to be published in final form later this year.

This paper will discuss the development of the human health effect assessment chapters of these documents.

Development of Water Quality Criteria

The objective of the human health effect assessment chapters of the EPA water quality criteria documents is to estimate ambient water concentrations of pollutants which do not represent a significant risk to the public. The assessments review all relevant information on individual chemicals or chemical classes in order to derive criteria which represent, in the case of suspect or proven carcinogens, various levels of incremental cancer risk, and in the case of other pollutants, "no observable effect levels".

Ideally, water quality criteria should represent levels for compounds in ambient water which would not pose a hazard to the human population. However, in any realistic assessment of human health hazard, a fundamental distinction must be made between absolute safety and recognition of some risk. Criteria for absolute safety would have to be based on detailed knowledge of dose/response relationships in humans including all sources of chemical exposure, the types of toxic effects elicited, the existence of thresholds for the toxic effects, the significance of toxicant interactions, and the variances of sensitivities and exposure levels within the human populations. In practice, such absolute criteria cannot be established because of deficiencies in both the available information and the means of interpreting this information. Consequently, the water quality criteria documents present human health effects criteria which minimize or specify the potential risk of adverse human effects due to substances in ambient water. Potential social or economic costs and benefits are not considered in the formulation of the criteria. Nor do the criteria recommendations consider technological limitations.

TYPES OF CRITERIA

Ambient water quality criteria are based on three types of biological endpoints: carcinogenicity, toxicity (i.e. all adverse effects other than cancer), and organoleptic effects.

For the purpose of deriving ambient water quality criteria, carcinogenicity is regarded as a non-threshold phenomenon. Using this assumption, "safe" or "no effect" levels for carcinogens cannot be established because even extremely small doses must be assumed to elicit a finite increase in the incidence of the response. Consequently, water quality criteria for carcinogens are presented as a range of concentrations associated with a corresponding range of incremental risk.

For compounds which do not display apparent carcinogenic activity, the threshold assumption is used in deriving a criterion. The threshold assumption is based on the premise that a reserve capacity exists which is thought to be depleted before clinical disease ensues. Alternatively, it may be assumed that the rate of damage will not be significant over the life span of the organism. Thus, ambient water quality criteria may be derived for non-carcinogenic chemicals, which will presumably result in no observable adverse effects in the exposed human population.

In some instances, criteria are based on organoleptic characteristics, i.e. thresholds for taste or odor. Such criteria are established when insufficient information is available on toxicologic effects or when the estimate of the maximum level of the pollutant in ambient water to protect from organoleptic effects is lower than the level calculated from toxicologic data. It is recognized that criteria based solely on organoleptic effects do not necessarily represent satisfactory approximations of low risk levels.

Several ambient water quality criteria documents cover classes of compounds, i.e. compounds which have varying degrees of structural similarity. Because prediction of biological effects based solely on structural parameters are difficult, an effort is made to derive a compound-specific criterion for each member of a class. A compound-specific criterion is defined as a level derived on each individual subject compound. For some chemical classes, however, a compound-specific criterion cannot be derived for each member of a class. In such instances, it is sometimes justifiable to derive a class criterion in which available data on one member of a class may be used to estimate criteria from other chemicals of the class even though a sufficient data base is not available for those compounds.

For some chemicals and chemical classes, the data base may be judged to be insufficient for the derivation of a criterion. In these cases, deficiencies in the available information are detailed.

APPROACH

The human health effects chapters of EPA's water quality criteria documents attempt to assess all information on the individual chemicals or classes of chemicals which might be useful in developing water quality criteria. Although primary emphasis is placed on identifying epidemiologic and toxicologic data, the assessments typically contain discussions on four topics: existing levels of human exposure, pharmacokinetics, toxicity, and criterion formulation.

For all documents, an attempt is made to include the known relevant information. Detailed literature searches and reviews are performed. Since each of the documents is prepared by a scientist who has either conducted original research on the compound or who has previously reviewed the toxicity of the compound, the probability of overlooking significant data is reduced. Review articles and reports are used for data evaluation and synthesis. Scientific judgment is exercised in reviewing and evaluating the data in each document and identifying the adverse effects for which protective criteria are sought. In addition, each of the documents is reviewed by a peer committee of scientists familiar with the compound under consideration. This committee evaluates the quality of the available data, the completeness of the data summary, and the validity of the derived criterion.

In the analysis and organization of the data, an attempt is made to be consistent in both format and the application of accepted scientific principles. The evaluation procedures used in the hazard assessment process follow the principles detailed by the National Academy of Sciences and guidelines of the EPA's Carcinogen Assessment Group.

EXPOSURE

The exposure section of the health effects chapters reviews known information on the current levels of human exposure to the individual pollutant from all sources. Much of the data is obtained from monitoring studies of air, water, food, soil, and human or animal tissue residues. The major purpose of this section is to provide background information on the contribution of exposure from water relative to all other sources. Consequently, this section is subdivided into exposure from ingestion in water, ingestion in food, inhalation, and dermal contact.

Information with regard to exposure is often valuable in developing a water quality criterion. Typically, a uniform exposure assumption is used in formulating the criteria which includes the consumption of contaminated drinking water and contaminated fish products. Exposure data, however, are useful for comparing the assumed intake to the expected contribution based upon available information. In addition, knowledge of exposure from all sources, not limited to drinking water and ingesting fish, can be used to justify the formulation of criteria based upon exposure to ambient water but recognizing contributions to total body intake from additional exposure routes.

The use of fish consumption as a typical exposure factor requires the quantification of pollutant residues in the edible portion of the ingested species. Bioconcentration factors are used to relate the expected pollutant residue in aquatic organisms to the pollutant concentration in the ambient waters in which they live. In order to estimate the average per capita intake of a pollutant due to consumption of contaminated fish and shellfish, a bioconcentration factor appropriate for the average U.S. diet is developed.

PHARMACOKINETICS

The pharmacokinetics section summarizes the available information on the absorption, distribution, metabolism, and elimination of the compound in humans and experimental mammals. Such information is useful in validation of inter- and intraspecies extrapolation processes, in estimating potential body burdens from long-term, low-level exposures, and in characterizing the mode of toxic action. Sufficient information on absorption and excretion in animals, together with a knowledge of ambient concentrations in food and water, could be useful in estimating body burdens of chemicals in the human population. Distribution data which determine target organs or tissues are needed for interspecies comparison. In terms of the quantitative derivation of criteria, pharmacokinetic data are crucial in cases involving estimates of equivalent oral doses from data on inhalation or other routes of exposure.

EFFECTS

The effects section summarizes the following types of information on both humans and experimental mammals:

Acute, subacute, and chronic toxicity
Synergistic or antagonistic action
Teratogenicity
Mutagenicity
Carcinogenicity

The major goals of this section are to assess the suitability of the data for use in establishing water quality criteria and to determine which biological endpoint, i.e., threshold, non-threshold, or organoleptic, should be selected for use in criterion formulation.

Because this section attempts to assess potential human health effects, data on documented human effects are actively sought. However, several problems with human epidemiological studies usually preclude the use of such data in generating water quality criteria. These problems, as summarized by the National Academy of Sciences are:

1. Epidemiology cannot tell what effects a material will have until after humans have been exposed. One must not conduct what might be hazardous experiments on man.

2. If exposure has been ubiquitous, it may be impossible to assess the effects of a material, because there is no unexposed control group. Statistics of morbidity obtained before use of a new material can sometimes be useful, but when latent periods are variable and times of introduction and removal of materials overlap, historical data on chronic effects are usually unsatisfactory.

3. It is usually difficult to determine doses in human exposures.

4. Usually, it is hard to identify small changes in common effects, which may nonetheless be important if the population is large.

5. Interactions in a "nature-designed" experiment usually cannot be controlled.

Although these problems often prevent the use of epidemiological data in quantitative risk estimates, qualitative similarities or differences between documented effects in humans and observed effects in experimental mammals are extremely useful in assessing the validity of animal-to-man extrapolations. Therefore, in each case an attempt is made to identify and utilize both epidemiologic and animal dose response data. Criteria derived from such a confirmed data base are considered to be the most reliable.

The decision to establish a criterion based on non-threshold effects is made by evaluating information on carcinogenicity, mutagenicity and teratogenicity. The first steps in the process of estimating risk are to determine whether the compound is likely to cause cancer in humans and to determine whether the data are adequate for the derivation of a water quality criterion. The criteria for the qualitative decision of carcinogenicity are outlined in the EPA's Cancer Guidelines and in the IRLG (Interagency Regulatory Liason Group) guidelines on carcinogenic risks. The assumption is made that a substance which induces a statistically significant response in animals by any route of exposure has the capability of causing cancer in humans from ingestion of this substance in water and edible aquatic organisms from that water. A chemical which has not induced a significant cancer response in humans or experimental animals is not identified as a possible carcinogen, even though its metabolites or close structural analogues might induce a carcinogenic response or even though they are mutagenic in in vitro systems.

It is recognized that some potential human carcinogens would not be identified by these rules. However, the derivation of a criterion concentration cannot be justified for them. There are other compounds for which there is plausible, but weak, qualitative evidence of carcinogenicity in experimental animal systems but which cannot be given a valid criterion concentration. This occurs in cases where the only evidence of animal or human carcinogenicity is obtained in a special carcinogen bioassay system such as mouse skin painting or strain A mouse pulmonary adenoma. The derivation of a criterion concentration for human consumption from these studies is considered invalid regardless of the qualitative outcome. In addition there are some compounds which are carcinogenic in humans via inhalation in one chemical form but have induced no response in animals or humans via ingestion of their soluble salts.

For those compounds which are not reported to induce carcinogenic effects or for those compounds on which carcinogenic effects data are lacking or insufficient, an attempt is made to define a "No Observable Adverse Effect Level (NOAEL)". In many respects, the evaluation of these studies is similar to that of carcinogenicity bioassays. In order to more closely approximate conditions of human exposure, preference is given to chronic studies involving oral exposures (dietary or in water) over a significant proportion of the organisms life span. Greatest confidence is placed in those studies which demonstrated dose related adverse effects as well as no effect levels. Considerable variability is generally encountered in the biological endpoints used to define NOAEL's which range from gross effects such as mortality to more subtle changes in biochemical, physiological, or pathological parameters. Given the variety of effects which some chemicals cause and the problems encountered in animal-to-man extrapolations, the derivation of a NOAEL is obtained using chronic data if available. Teratogenicity, reproductive impairment, and behavioral effects may be significant consequences of environmental contamination. In those cases where carcinogenic or other chronic effects occurred at exposure levels below those causing teratogenicity, reproductive impairment, or behavioral effects, the former is used in deriving the criterion. For most of the compounds evaluated thus far by EPA, teratogenicity and reproductive impairment occur at doses near maximum tolerated levels with administration schedules which do not reasonably approximate environmental exposures. Information on behavioral effects, which is of considerable potential significance, is not available on most of the compounds under study. Consequently, most NOAEL's derived from chronic studies are based either on gross toxic effects or on effects which can be directly related to functional impairment or defined pathological lesions.

For compounds on which adequate chronic toxicity studies are not available, studies on acute and subacute toxicity assume greater significance. Acute toxicity studies usually involve single exposures at lethal or near lethal doses. Subacute studies usually involve exposures over about 10% of the life span of the test organisms, e.g., 90 days for the rat with an average life span of 30 months. Such studies are useful in establishing the nature of the toxic effects, including the target organs, metabolic behavior, physiological/biochemical effects, and patterns of retention and tissue distribution.

In some cases where adequate data are not available from studies using oral routes of administration, NOAEL's for oral exposures are estimated from dermal or inhalation studies. Such estimates involve approximations of the total dose based on assumptions about breathing rates and/or magnitude of absorption.

CRITERION RATIONALE

The criterion rationale section reviews existing standards for the chemical, summarizes data on current levels of human exposure, attempts to identify special groups at risk, and defines the basis for the recommended criterion.

Information on existing standards is included primarily for comparison with the proposed water quality criteria. Some of these standards, such as those recommended by the Occupational Safety and Health Administration (OSHA) or the American Conference of Governmental Industrial Hygienists (ACGIH), are based on toxicological data but are intended as acceptable levels of occupational rather than environmental exposure. Other levels, such as those recommended by the National Academy of Sciences in its publication on Drinking Water and Health or in the EPA Primary Drinking Water Standards are more comparable to the proposed water quality criteria. Emphasis is placed on detailing the bases for the existing standards, wherever possible.

Summaries of current levels of human exposure presented in this section specifically address the applicability of the data to generating water quality criteria. The identification of special groups at risk - either because of geographical or occupational differences in exposure or biological differences in susceptibility to the compound - focus on the impact that these groups should have on the development of water quality criteria.

In the section on the basis for the recommended criteria all of the data used in developing the criteria are summarized and, if necessary, qualified.

GUIDELINES FOR CRITERIA DERIVATION

The derivation of water quality criteria from data on laboratory animal toxicity tests is essentially a two step procedure. First, a total daily intake for humans must be estimated which establishes either a defined level of risk for non-threshold effects or a no-effect level for threshold effects. Secondly, some assumptions must be made about the contribution of contaminated water and fish/shellfish consumption to the total daily intake of the chemical. These assumptions may then be used to calculate the water quality criterion as it relates to the tolerable daily intake.

A. Non-Threshold Effects

After the decision has been made that the compound has the potential of causing cancer in humans and that data exist which permit the development of a criterion, the water concentrations which are estimated to cause a lifetime carcinogenic risk of 10^{-5}, 10^{-6}, 10^{-7} are determined. These concentrations are calculated by fitting the available data to a dose-response model. The basic EPA Carcinogen Assessment Group (CAG) dose response model has been to this time the "one-hit" model. Because this model approximates a straight line response through the origin at low doses, it is sometimes called a linear, no-threshold, dose-response model. It states that any exposure to a carcinogen, however small, results in some chance of cancer occurrence, and the probability (or the risk) of getting cancer from low exposure increases linearly with the dose.

This model has been used for risk estimations because it is consistent with three basic concepts in chemical carcinogenesis: a) the dose-response curve for mutagenesis in bacterial systems is linear with no threshold for both radiation- and chemical-induced damage; b) there are both theoretical reasons, based on valid concepts of the mechanisms of chemical carcinogensis, as well as a wealth of data, which indicate that chemicals which cause mutations are likely to induce cancer in animals and therefore presumably in people; and c) epidemiology studies in human populations exposed to animal carcinogens (radiation, aflatoxin, cigarette smoke) show an incidence-exposure relationship that is linear down to very small doses with no evidence of a threshold.

The EPA Carcinogen Assessment Group has very recently decided to use a multi-stage model for low-dose risk extrapolation from animal carcinogenesis data in the final water quality criteria documents. The reasons for this change are that the new procedure is more systematic, it invokes fewer arbitrary assumptions, and it incorporates data from all of the dose groups which are consistent with the multi-stage model. At the same time it is conceptually consistent with the linear, no-threshold concept on which the one-hit procedure was based.

The entire procedure, both in the choice of the model and in the selection of which of several sets of carcinogenic data to use in the model, is designed to determine roughly how severe the carcinogenic hazard could be if the chemical does have the potential of causing cancer in humans. The hazard is not likely to be higher but could be lower if some threshold or non-linear response exists. Therefore the water quality criteria concentrations derived from the procedure are intended to be estimates of the smallest concentrations that are likely to give rise to the risk indicated.

After determining that a substance is carcinogenic and that the existing information is adequate for developing a water quality criterion, a choice must be made of which of the data sets from several studies to use in the model. For some chemicals, several studies in different animals species and strains, each run at several doses and possible different routes of exposure, are available. It is also necessary to correct for metabolism differences between species and absorption factors via different routes. The procedures used in evaluating these data are consistent with the approach of making a conservative risk estimate.

B. THRESHOLD EFFECTS

In developing guidelines for deriving criteria based on non-carcinogenic responses, adverse effects are defined as any effects which result in functional impairment and /or pathological lesions which may affect the performance of the whole organism, or which contribute to a reduced ability to respond to an additional challenge.

In using this approach, the selection and justification of uncertainty factors are critical. The basic definition and guidelines for using uncertainty factors has been given by the National Academy of Sciences. A safety factor or uncertainty factor is defined as a number that reflects the degree or amount of uncertainty that must be considered when experimental data in animals are extrapolated to man. When the quality and quantity of experimental data are satisfactory, a low uncertainty factor is used; when data is judged to be inadequate or equivocal, a larger uncertainty factor is used.

The following general guidelines have been adopted in establishing the uncertainty factors:

1. Valid experimental results from studies on prolonged ingestion by man, with no indication of carcinogenicity.
   Uncertainty Factor = 10

2. Experimental results of studies of human ingestion not available or scanty (e.g., acute exposure only). Valid results of long-term feeding studies on experimental animals or in the absence of human studies, valid animal studies on one or more species. No indication of carcinogenicity.
   Uncertainty Factor = 100

3. No long-term or acute human data. Scanty results on experimental animals. No indication of carcinogenicity.
   Uncertainty Factor = 1,000

A considerable judgement must be used to justify the application of the safety factors in deriving the criteria. In those cases where the data do not completely fulfill the conditions for one category of uncertainty and appear to be intermediate between two categories, a more restrictive safety factor or, more appropriate, an intermediate operative uncertainty factor is used.

In the determination of the appropriate use of the uncertainty factors "no indication of carcinogenicity" is interpreted as the absence of carcinogenic data from animal experimental studies or human epidemiology. Available short-term carcinogenicity screening tests are reported in the criteria documents, but they are not used either for derivation of numerical criteria nor to rule out the uncertainty factor approach.

Because of the high degree of judgement involved in the selection of a safety factor, the criterion derivation section of each document provides a detailed discussion and justification for both the selection of the safety factor and the data to which it is applied. This discussion reflects a critical review of the available data base. A number of factors are considered including number of animals, species, and parameters tested; quality of controls; dose levels; route; and dosing schedules. A valid effort is made to differentiate between results which form a toxicologically sufficient data base and data which may be equivocal in nature.

Unlike criteria for carcinogens, in which the assumption of low dose linearity precludes the necessity for defining total exposure in the estimation of increased incremental risk, for non-carcinogens, Allowable Daily Intakes (ADIs) and criteria derived therefrom are calculated from total exposure data which include contribution from diet and the environmental media. In terms of scientific validity, the accurate estimate of the Allowable Daily Intake is the major factor in satisfactory derivation of water quality criteria.

Estimating equivalencies of dose-response relationships from one route of exposure to another introduces an additional uncertainty in the derivation of criteria. Consequently, whenever possible, ambient water quality criteria should be based on data involving oral exposures. If oral data are insufficient, data from other routes of exposure may be useful in criterion derivation process.

Inhalation data, including TLV's or similar values, are most common alternative to oral data. Estimates of equivalent doses can be made on the basis of availability of pharmacokinetic data for oral and inhalation routes, on the basis of measurements of absorption efficiency from ingested or inhaled chemical, or on the basis of comparative excretion data when the metabolic pathways can be established to be equivalent after oral or inhalation dosing. When sufficient pharmacokinetic data are available, the use of accepted pharmacokinetic models provides the most satisfactory approach for dose conversions. However, if available pharmacokinetic data are marginal or of questionable quality, pharmacokinetic modeling is inappropriate.

Threshold Limit Values (TLV's) are established by the American Conference of Governmental Industrial Hygienists (ACGIH) and represent 8 hour time weighted average concentrations in air intended to protect workers from various adverse health effects over a normal working lifetime. Similar values are set by NIOSH (criteria) and OSHA (standards) for 10 and 8 hours exposures, respectively. To the extent that these values are based on sound toxicologic assessments and have been protective in the work environment, they provide useful information for deriving or evaluating water quality criteria. However, each TLV must be carefully examined to determine if the basis of the TLV contains data which can be used directly to derive a water quality criterion using the uncertainty factor approach. In addition, the history of each TLV should be examined to assess the extent to which it has been protective of worker safety. In each case, the types of effects which TLV's are designed to protect against should be examined in terms of their relevance to exposure in water. For the establishment of equivalency of effect, it needs to be demonstrated that the chemical is not an irritant at the site of entry and that there is no significant first pass effect for one of the routes of exposure (i.e., oral or inhalation).

If the TLV or similar value is recommended as the basis of the criterion, a consideration of the above points is explicitly stated in the criterion derivation section of the document. Particular emphasis is placed on the quality of the TLV relative to the available toxicity data which should be given preference over TLV's or similar established values unless a compelling case can be made against it. If the TLV can be justified as the basis for the criterion, then the problems in estimating acceptable oral doses from inhalation data must be addressed.

C. ORGANOLEPTIC CRITERIA

Organoleptic criteria define concentrations of materials which impart undesirable taste and/or odor to water. In developing and utilizing such criteria two factors must be appreciated: the limitations of most organoleptic data and the human health significance of organoleptic properties.

The publications which report taste and odor thresholds are with very few exceptions cryptic in their descriptions of test methodologies, number of subjects tested, concentration:response relationships, and sensory characteristics at specific concentrations above the threshold. Thus the quality of the data is usually significantly poorer than the toxicological data used for the setting of other criteria. Consequently, a clear critical evaluation of the available data on a compound's organoleptic characteristics should appear in the criteria document.

Organoleptic criteria are not based on toxicologic information and have no direct relationship to potential adverse human health effects. Although sufficiently intense organoleptic characteristics could result in depressed fluid intake which could aggravate kidney and circulatory disease, such effects should not be assumed in assessing the significance of organoleptic criteria. Consequently, unless data are available suggesting an indirect human health effect through fluid consumption, organoleptic data form the basis of criteria only based on aesthetic consideration.

ESSENTIAL ELEMENTS

Some chemicals, particularly certain metals, may be essential to biological organisms at low levels but may be toxic and/or carcinogenic at high levels. Because of potential toxic effects, it is legitimate to establish criteria for such essential elements. However, criteria must consider essentiality and cannot be established at levels which would result in deficiency of the element in the human population.

Elements are accepted as essential if listed by the NAS Food and Nutrition Board or a comparably qualified panel. Elements not yet determined to be essential but for which supportive data on essentiality exists, need to be reviewed by such a panel.

To modify the toxicity and carcinogenicity based criteria, essentiality must be quantified either as a "recommended daily allowance" (RDA) or "minimum daily requirement" (MDR). These levels are then compared to estimated daily doses associated with the adverse effect of primary concern. The difference between the RDA/MDR and the daily doses causing a specified risk level for carcinogens or ADI's for non-carcinogens defines the spread of daily doses from which the criterion is derived. Because errors are inherent in defining both essential and maximum tolerable levels, the criterion is derived from dose levels near the center of such a spread. The decision to use either the MDR or RDA is guided by the spread of the doses and the quality of the essentiality and toxicity estimates.

The modification of criteria by consideration of essentiality must take into account all routes of exposure. If water is a significant source of the MDR or RDA, the criterion must allow for attainment of essential intake. Conversely, even when essentility may be attained from non-water sources, standard criteria derivation methods may be adjusted if the derived criterion represents a small fraction of the ADI or MDR. On a case by case basis, the modification of standard criteria derivation methods may include the use of different safety factors for non-carcinogens or other modifications which can be explicitly justified.

USE OF EXISTING STANDARDS

For some chemicals for which criteria are to be established, drinking water standards already exist. These standards represent not only a critical assessment of literature, but also a body of human experience since their promulgation. Therefore, it is valid to accept the existing standard unless there is compelling evidence to the contrary. This decision should be made after considering the existing standards vs. new scientific evidence which has been accumulated since the standards have been promulgated.

CARCINOGEN-BASED CRITERIA: ASSESSMENT OF UNCERTAINTIES

Kenny S. Crump

Science Research Systems, Inc.
P. O. Box 7356
Ruston, Louisiana 71270

ABSTRACT

Sources of uncertainty in the calculation of carcinogen-based water quality criteria include uncertainty in extrapolating from high doses to low doses, uncertainty in extrapolating across species, uncertainty related to the value of the bioaccumulation factor in fish and uncertainty resulting from variable quality of the carcinogenicity data upon which the criteria are based. It is suggested that the degree of uncertainty related to data quality can be communicated through the assigning of indices to the data bases used in determining water quality criteria. Two such indices are suggested; an index which would reflect the likelihood that the chemical is a human carcinogen and a Risk Assessment Confidence Index (RACI) which would reflect the overall extent and quality of the data for quantitative estimation of carcinogenic potency in humans. Specific recommendations are made for calculating a RACI.

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INTRODUCTION

A carcinogen-based water quality criterion is supposed to correspond to a concentration level such that if humans throughout their life drink water contaminated at a level below the criterion level and also eat fish taken from such water, then the additional lifetime risk of cancer from ingestion of the chemical from these sources will be less than a specified low amount, such as 10^{-5}, 10^{-6}, or 10^{-7}. Most of the criteria calculated thus far by EPA have relied upon animal bioassay data. There are several sources of uncertainty related to the calculation of such criteria, including:

1. Uncertainty in the estimation of low-dose effects from high dose data.
2. Uncertainty resulting from the quantification of effects in humans based upon observed effects in animals,
3. Uncertainty resulting from the quality of the carcinogenicity data upon which the criterion is based,
4. Uncertainty related to the value of the bioaccumulation factor in fish.

Except for uncertainty related to extrapolation across species (2 above), these areas of uncertainty also apply to criteria calculated from human epidemiological data. However, epidemiological data involve uncertainties of their own, particularly those related to possible confounding factors and the exposure levels to which the study cohort was exposed.

Usual statistical procedures for quantifying the uncertainty of estimates, e.g., the use of statistical confidence limits, require the assumption that the mathematical model used is correct. This assumption is particularly crucial whenever estimates are made outside the range of observation. A major source of uncertainty in carcinogenic risk estimation stems from the uncertainty regarding the appropriate model to use when extrapolating risks from high dose to low doses. Such "model uncertainty" is not amenable to quantification using statistical confidence limits. In fact, the decision to extrapolate from high doses to estimate low dose risks, rather than estimating such risks by direct experimentation at the dose levels of interest, involves an implicit decision to trade off statistical uncertainty in exchange for model uncertainty. Additionally, it is usually necessary to use animal data to estimate the carcinogenic potency of a chemical in humans. The animals are sometimes exposed by a different route and possibly through a different temporal dose regimen than the target human population. Thus, in all of these dimensions - dose-range, species, and sometimes exposure route and temporal exposure pattern - one is attempting to make estimates outside the observable range. Although confidence limits based upon the statistical variation in results from animal bioassays may be of value in the setting of water quality criteria, they do not reflect all of these areas of uncertainty.

I wish to address here particularly the third area of uncertainty mentioned earlier - the uncertainty due to the quality of the carcinogenicity data - and to discuss ways of quantifying and communicating this uncertainty. There are recognizable degrees of uncertainty associated with data quality, even though the degrees may be difficult to quantify. For example, we should have more confidence in a water quality criterion calculated from an experiment in which animals were exposed through water than one in which exposure was by inhalation.

One possible method for communicating such degrees of uncertainty is through the use of indices. As an example of such an index, along with a water criterion might be furnished an index which reflects the strength of the evidence that the chemical is a human carcinogen. Such an index could be called a Human Carcinogenicity Index (HCI). An HCI value of 4 might, for example, represent "good" epidemiological evidence with corroborative animal evidence. An HCI of 3 might represent strong evidence of carcinogenicity in at least two animal species, and so on. Of course, in defining such an index, we would want to make rather explicit definitions of terms such as "strong evidence", etc. Note that such an index does not necessarily reflect the extent of the data since a low HCI might be either due to the fact that the chemical has not been adequately studied, or else due to the fact that the evidence for carcinogenicity is truly equivocal even after extensive study.

Another possible index is one which indicates the extent and quality of the data for calculating a quantitative carcinogen-based water quality criterion. Such an index might be labeled a Risk Assessment Confidence Index (RACI).

Although a RACI would be somewhat similar to an HCI, the two indices would have different purposes and reflect different features of the data. For example, whereas a HCI would depend upon the carcinogenic responses in an experiment, a RACI would essentially depend only upon experimental protocol.

If the data on a particular chemical had a high HCI and also a high RACI, then the criterion calculated from this data would have a firm basis, as far as the data are concerned. If a data base had a high HCI and a low RACI, then, although the chemical would be likely to be a human carcinogen, the data would be weak for calculating a numerical criterion. In this situation, it would be not unlikely that data from a new study would require a major change in the numerical value of the criterion.

Such indices as these would provide policy makers with easily assimilated measures of the strength of the data base upon which their decision or action is based. Given only water quality criteria numbers, regulators may have little feeling for the quality of the data upon which the criteria are based. Although such information may be available in prose form in the original criteria documents, whenever a sizeable number of chemicals are involved, it may be difficult to extract and assimilate the necessary information. Further, these documents are written by different individuals and the standards of different authors may not be comparable; some authors will be more cautious than others in their evaluation of data or criticism of protocols. On the other hand, proper use of indices which have a clearly defined basis could lead to a more balanced view of data quality.

Additional benefits might also accrue from the use of indexing schemes. A RACI might be used, for example, to determine the most appropriate data base for calculating a water quality criterion whenever data are available from experiments employing a variety of protocols. The formal process of calculating such indices should increase awareness of the relative merits of different categories of data and of the areas in which improvements in data quality are needed.

In the remainder of the paper, I wish to suggest a tenative approach for calculation of an index (RACI) to reflect the extent and quality of a data base for quantitative calculation of water quality criteria. The procedure which I shall outline utilizes only animal bioassay data. However, it can be easily extended to utilize epidemiological data as well (Crump and Howe[1]). The formulation of the method reflects judgements regarding which items in an experimental protocol are important to quantitative risk assessment and the relative importance of these items.

A different approach to quantifying uncertainties in carcinogenic risk estimation has recently been proposed by Crouch and Wilson[2]. They suggest a procedure for incorporating three types of uncertainty - the uncertainty in the estimation of carcinogenic potency in animals, the uncertainty in the interspecies conversion factor used to convert animal risks to human risks, and the uncertainty in the exposure of the target human population to the carcinogen - into a single number which they interpret as an estimate of the variance of the calculated carcinogenic risk.

PROCEDURE FOR ASSIGNING A RISK ASSESSMENT CONFIDENCE LIMIT (RACI)

There are several stages to assigning an index to a complete data set. First indices are assigned to individual bioassay and then combined to form the overall RACI. In assigning an index to an individual bioassay, ranks are first assigned to various items which are judged to be important in a quantitative assessment of risk. These individual ranks are then used as input to a specific formula which yields the RACI for the bioassay.

The output of the indexing scheme is then an index for each bioassay and also an index for the complete data base. Higher values of an index indicate more extensive and higher quality data. An index for a single bioassay ranges between zero and 5. An index for a complete data base can be as small as zero and has no definite upper limit.

The scheme for assigning a RACI is recommended as a guide and not as an inflexible rule. It is impossible to foresee and adequately account for all possible circumstances surrounding an animal bioassay which might affect the quantitative predictive value of the data. Thus, unusual circumstances, such as failure to follow standard laboratory practices, should be considered on a case-by-case basis. Nevertheless, to be of value, RACIs calculated by different groups or at different times must be comparable. To promote uniformity of results, I have attempted to make the definition of a RACI as explicit as possible even though this required that some fairly arbitrary decisions be made regarding the ranking of individual items.

RANKING OF INDIVIDUAL ITEMS IN AN ANIMAL BIOASSAY

Listed below are the items to be ranked and the procedure for assigning a rank for each item. Each item may be assigned either two or three possible ranks, with higher ranks indicating a higher level of appropriateness for quantitative risk assessment.

A. Temporal Exposure Pattern

3. Exposure beginning at birth or during fetal development (or, for estimation of risks to occupationally exposed populations, exposure beginning at weaning or soon thereafter) and continuing until the end of the experiment.

2. Exposure beginning at weaning or soon thereafter and continuing until at least 85% of the average lifespan of the experimental animals has elapsed.

1. Any temporal exposure pattern which does not qualify for A.2 or A.3, but for which a positive carcinogenic response is observed.

B. Animal Lifespan

3. Average lifespans at least 95% of the normal average species lifespan for the control group and either each entire treatment group or each group of tumor-free animals from each treatment group.

2. Average lifespans at least 80% of the normal species lifespan for the control group and either each entire treatment group or each group of tumor-free animals from each treatment group.

1. Standard in B.2 is not met, but a significant carcinogenic response is realized.

C. Identity of Test Material

2. Test material to which animals are exposed identical to that for which human risk is to be estimated.

1. Test material to which animals exposed similar chemically and physically to the material for which risk estimates are required

in humans and there is reason to believe that the two materials may produce a similar carcinogenic response.

D. Number of Treatment Groups

3. Control group plus at least four treatment groups with one group exposed to a maximum tolerated dose (MTD).

2. Control group plus at least two treatment groups with one group exposed to an MTD.

1. Control group plus a single treatment group.

E. Exposure Route

3. Exposure is through water or food.

2. Feeding by stomach tube or certain other routes where animal and human distribution and metabolism seem comparable.

1. Other routes of exposure not covered in E.3 or E.2 but which resulted in a positive carcinogenic effect in animals.

F. Pathology

3. Pathology procedures comparable to those recommended in the NCI Carcinogen Bioassay Guidelines.[3]

2. Pathology procedures of comparable quality to those recommended in the NCI Carcinogen Bioassay Guidelines, but applied to only a few target tissues.

1. Pathology procedures below standards recommended in the NCI Carcinogen Bioassay Guidelines, but which are applied consistently to both treatment and control groups and through which a positive carcinogenic effect was discovered.

G. Size of Experiment

3. Greater than 500 total animals.

2. Between 100 and 500 animals.

1. Less than 100 total animals and a positive carcinogenic effect was detected.

H. Time-to-tumor Data

2. Time-to-tumor data available.

1. Time-to-tumor data unavailable.

I. Sexes Represented

2. Data available on both sexes.

1. Data available on a single sex only.

The choices of items and the ranking of these items represents judgements regarding which features in an animal bioassay are important to quantitative assessment. Several of the items, such as Pathology, Size of Experiment, Sexes Represented and Number of Treatment Groups relate to the thoroughness with which the study was conducted. The ranking of other items reflects the judgement that quantitative predictions in humans from animal bioassays are least uncertain whenever circumstances of dose and lifespan in the test species mimic those of the target human population to the fullest extent possible. This means that since we wish to estimate lifetime risks in humans, we prefer an animal experiment in which animals live out their complete usual lifespans without early sacrifices. Similarly, to estimate risks from lifetime exposure through drinking water, it is preferable for exposures in animals to be through water or food, to begin at birth and continue until the end of the experiment.

PROCEDURES FOR ASSIGNING A RACI TO AN ANIMAL BIOASSAY

Table 1 is used for obtaining an overall index (RACI) between 0 and 5 for an experiment using the rankings of the individual items of the experiment. The RACI of an experiment is the highest overall index for which all of the individual ranks meet or exceed the levels associated with the RACI. For example, if the ranks of the individual attributes of an experiment are A-3, B-2, C-2, D-2, E-3, F-2, G-2, H-2, I-1, the experiment would be assigned a RACI of 3. If B-2 were changed to B-1, the RACI would change from 3 to 1.

The assignment of RACIs by use of Table 1 represents judgements regarding the relative effects of the various factors in an experimental protocol upon the validity of a risk assessment. Identity of the Test Material, Pathology, and Exposure Route were regarded as being the most critical items. Correspondingly, if any one of these items receives

TABLE 1

CHART FOR OBTAINING A RACI FOR EXPERIMENTAL ANIMAL DATA FROM RANKS OF INDIVIDUAL ITEMS

	RANKS OF INDIVIDUAL EXPERIMENTAL ITEMS								
OVERALL INDEX OF EXPERIMENTAL DATA	TEMPORAL EXPOSURE PATTERN	ANIMAL LIFESPAN	IDENTITY OF TEST MATERIAL	NUMBER OF TREATMENT GROUPS	EXPOSURE ROUTE	PATHOLOGY	SIZE OF EXPERIMENT	TIME-TO-TUMOR DATA	SEXES REPRESENTED
	A	B	C	D	E	F	G	H	I
5	3	3	2	3	3	3	3	2	2
4	2	3	2	2	3	3	2	1	2
3	2	2	2	2	3	2	2	1	1
2	1	2	2	2	2	2	1	1	1
1	1	1	2	1	2	2	1	1	1
0	1	1	1	1	1	1	1	1	1

a rank of one, the overall experiment is assigned a RACI of zero.

The indexing scheme is based upon the premise that high ranks for some items cannot compensate for deficiencies of other items. Thus, for example, if the exposure pattern item ranks no higher than one, the overall experiment can have an index no greater than two, regardless of the ranks assigned to the other items.

PROCEDURE FOR ASSIGNING A RACI TO AN OVERALL DATA BASE

Thus far, we have described how to assign a RACI to an individual bioassay involving only one species. Whenever data are available from more than one study on the same strain in a particular species, it is recommended that a RACI relative to the species be defined as the maximum of the RACIs of the individual studies. Whenever data are available on two or more separate species, it is recommended that the RACIs for each species be summed to yield a RACI for the overall data base.

The choice of the maximum of the RACIs within a species as the species-specific RACI reflects the judgement that a poorer study adds very little, if any, additional strength to the data base already provided by a better study using the same species. This does not necessarily hold, however, for two studies of different qualities on different species. Even a relatively poor study may reveal significant information regarding the carcinogenic potential of the test chemical that would not be available from a better study on a different species.

An intermediate approach is called for whenever data are available on two different strains of the same species. We may wish to assign a higher RACI to the species than the maximum of the individual RACIs, but not as large as the sum of the RACIs. The exact procedure to follow might best be decided on a strain-by-strain basis.

EXAMPLE OF THE CALCULATION OF A RACI

To illustrate the calculation of a RACI, we applied the indexing scheme to the NCI[4] Bioassay data on the chemical diazinon. The result is given in Table 2. The study involved both mice and rats and the species-specific RACI for each was four. Thus, the overall data were assigned a RACI of eight.

The maximum RACI that could be assigned to such a study is ten. To achieve a RACI of ten, the study would have to involve at least 500 animals of each species, have an exposure which began at birth instead of at weaning, and have at least four treatment groups. Since NCI bioassays typically do not satisfy any of these conditions, an NCI study involving both rats and mice will generally have a RACI no larger than eight. The RACI for an NCI bioassay could be less than eight, if, for example, there were early deaths in the experiment which were unrelated to carcinogenesis.

TABLE 2

EXAMPLE CALCULATION OF A RACI

SUBSTANCE: DIAZINON
SPECIES: B6c3F1 mice and
Fisher F344 rats

SOURCE: NCI-CG-TR-137 HEW

		RANKS	
		RAT	MOUSE
A.	TEMPORAL EXPOSURE PATTERN	2	2
B.	ANIMAL LIFESPAN	3	3
C.	IDENTITY OF TEST MATERIAL	2	2
D.	NUMBER OF TREATMENT GROUPS	2	2
E.	EXPOSURE ROUTE	3	3
F.	PATHOLOGY	3	3
G.	SIZE OF STUDY	2	2
H.	TIME-TO-TUMOR DATA	2	2
I.	SEXES REPRESENTED	2	2
	SPECIES SPECIFIC RACIs:	4	4
	OVERALL RACI:	8	

LIMITING ITEMS: Size of Study
Temporal Exposure Pattern
Number of Treatment Groups

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HIGH RISK GROUPS: A CRITICAL FACTOR IN THE DEVELOPMENT OF ENVIRONMENTAL HEALTH CRITERIA AND STANDARDS

Dr. Edward J. Calabrese

School of Health Sciences
Division of Public Health
University of Massachusetts
Amherst, MA 01003

ABSTRACT

There is a growing recognition that there is a high degree of variability in the response of humans to different exposure levels of air and water pollutants. Knowledge of those groups considered at high risk to pollutants should play a critical role in the assessment of what is a acceptable level of pollutant exposure. While EPA has recognized and included in the knowledge of high risk groups in the standard setting process, further appreciation and development of this concept can play an important role in establishing acceptable levels of exposure to carcinogens and in the development of cost/benefit assessments.

ACKNOWLEDGEMENTS

This work was based on a previous paper entitled, "The Role of High Risk Groups in the Derivation of Environmental Health Standards" which was previously presented at the Annual American Chemical Society Meetings in September of 1979 in Washington, DC and is being subsequently published within Reviews in Environmental Health, 1980 (In Press). The present paper has extended the previous version by assessing the role of high risk groups in the derivation of risk assessments via linear biostatistical models and by a consideration of the health implications for high risk groups in the use of recreational water as a community drinking water source. Finally, while the previous paper dealt with both air and drinking water standards, the present paper almost exclusively emphasizes the derivation of drinking water standards.

INTRODUCTION

Numerous studies have indicated that there is a high degree of variability in the response of humans to different exposure levels of air and water pollutants [1-4]. In fact, there is a highly variable dose-response relationship in a heterogeneous population such that at each level of pollutant exposure, depending on the adaptive capacity of the specific individual considered, there is likely to be an increase in the toxic response including prolongation of illness, and in certain cases, premature death. Perhaps the most critical question is not what is a "safe" numerical standard, but how many individuals are adversely affected at each successive level of exposure [5]. Because of the need to establish more accurate cost/benefit relationships with regard to proposed health standards, it is necessary to provide decision-makers with information which will define the responses of humans to the toxic substances in question at each successive level of exposure.

Knowing which groups of individuals are at high risk with respect to pollutants is very important in answering this question since these are the people who will be the first to experience morbidity and mortality as pollutant levels increase. If the high risk segments are protected, then the entire population is also protected. Information concerning both the identification and quantification of high risk groups should play an integral role in the derivation of environmental health standards. Consequently, it is the intention of this paper to examine the role of high risk groups in the derivation of environmental health standards with particular emphasis on how the U.S. EPA has attempted to utilize this knowledge in setting drinking water standards.

THE SAFETY-FACTOR CONCEPT

In trying to assess the potential role of high risk groups in the derivation of environmental health standards, it is necessary to consider to what extent EPA has utilized the concept of high risk groups within the standard setting process. Perhaps the most common approach utilized by EPA in setting exposure standards for noncarcinogens has been by the implementation of safety factors. Blanket-safety factors of any designated number (e.g. 10, 100, 500 etc.) implicitly recognize that certain people are more sensitive to pollutants than others. However, they are inherently imprecise. The precise difference in sensitivity between a statistically "normal" individual and groups at increased risk will vary for the different causes of the high risk condition. Thus, wherever safety factors have been applied it suggests that high risk groups have been taken into consideration, but only in a generalized and non-specific manner.

a. Noncarcinogenic Chlorinated Hydrocarbon Insecticides and Herbicides

There are numerous examples where EPA has utilized this technique in attempting to deal with protecting the high risk individuals. Take, for example, the national drinking water standards for non-carcinogenic chlorinated hydrocarbon insecticides and herbicides [6]. These substances were reported to have been tested in two animal species, the rat and dog. Chronic toxicity testing revealed the lowest level of pollutant that the animal could ingest with either minimal or no toxic effects. Once this had determined, the policy-makers selected that species which was the most sensitive to the substances in question. If there were no supporting human exposure data to consider then a safety factor of 500 was applied to the minimally toxic dose in the most sensitive of the two species of animals (i.e. the minimally toxic dose was divided by 500). This number was taken to be the total amount of insecticide (or herbicide) to which a human could be exposed each day over an unspecified period of time without suffering any adverse health effects. Next, amounts of the chlorinated hydrocarbons normally consumed via the diet were derived from market basket surveys. If this amount was substantially less then 1/500 of a known toxic dose in the most sensitive animal species tested (i.e., the assumed safe level of intake for humans), then a drinking water limit was established which would permit 20% of the safe limit to be consumed via water. Several questions naturally arise when evaluating such a methodological scheme. For example, why was 500 chosen as a safety factor? What assurances exist that it would provide sufficient protection for the general population as well as high risk groups? Who, in fact, are the groups considered at increased risk? On the other hand, does this approach result in unnecessarily strict standards?

b. Barium

Another example in which this generalized safety factor procedure was used involved the EPA drinking water standard for barium. Stokinger and Woodward [7] presented a theoretical attempt at deriving a drinking water standard for barium based entirely on the industrial threshold limit value (TLV) for barium of 0.5 mg/m^3 air. According to Stokinger and Woodward, TLVs for industrial chemicals in air can be reasonably adapted to drinking water standards if it is recognized that the TLV values offer safety to individuals who are exposed to the toxic substance for 8 hrs./day, 40 hrs/ week over a working lifetime. TLVs are not designed to insure the protection of all individuals. That is, some of the hypersusceptible segments of the population may be adversely affected. In contrast, community drinking water standards are designed to protect all of the citizens including the segments of the population at high risk, for their entire lives. Without going into the details of this TLV derivation technique, suffice it to say that once a reasonably safe limit was derived for the healthy adult population, they considered in a general, non-specific manner, the health of potentially hypersusceptible segments of the population. Thus, the calculated "safe" exposure level for a normal adult population of 2 ppm was lowered by 50% of its value (i.e. 1 ppm) to account for those who may be at increased risk. This procedure of utilizing the industrial TLV for the development of the barium drinking water standard was not only officially accepted to be one of the PHS standards of 1962, but is essentially reported by EPA for their justification of the present national standard for barium. Once again, there was the use of a safety factor designed to prevent the development of adverse health effects in the most susceptible segments of the population.

The main problem with such an approach is its lack of specificity. No attempts were made to identify those at increased risk, to what extent they would be more susceptible and, how many of these hypersusceptibles need to be protected. These statements are not necessarily intended to be critical of "safety factors" or the individuals who apply them. In fact, when only limited data are available, imprecise safety factors may be the only realistic options available. These comments are intended, however, to illustrate deficiencies in certain approaches to the problem of risk assessment.

SPECIFIC HIGH RISK GROUP CONSIDERATION

The drinking water standard for nitrate of 10 mg NO_3 as N per liter is designed to prevent the formation of elevated levels of methemoglobin (MetHb) in infants. In the presence of nitrite, hemoglobin is oxidized to MetHb, which is not able to reversibly combine with oxygen. Levels of 1 to 2% and 2 to 5% MetHb are typical in the blood of adults and infants, respectively. When concentrations are less then 5% MetHb, there are no obvious indications of toxicity. However, when levels of MetHb increase from 5 to 10%, clinical signs of toxicity may appear [4].

Infants are at considerable risk to nitrate related toxicity as compared to adults. Factors that predispose infants to the development of MetHb formation include:

1. the infant has an incompletely developed ability to secrete gastric acid; this permits the gastric pH to be high enough (5-7 pH) to allow nitrate-reducing bacteria to grow high in the gastrointestinal tract and thereby convert the nitrate to nitrite before absorption into the circulation [8].
2. infants are born with fetal hemoglobin; this form of hemoglobin is more susceptible than adult hemoglobin to be oxidized to MetHb [9].
3. infants have a somewhat diminished enzymatic capability to reduce MetHb to hemoglobin [10].

Research has revealed that as the level of nitrate increased beyond 20 ppm there is a marked upshift in the frequency of infant methemoglobinemia but not among adults [6]. Consequently, a standard of 10 ppm is principally designed to prevent the occurrence of elevated levels of MetHb in infants.

EXAMPLE WHERE EPA REJECTED PROTECTION OF HIGH RISK GROUPS

Studies with rats have revealed that when the concentration of cadmium reaches 200 ppm in the kidney [11], renal damage will be initiated. EPA calculated that for humans to reach a level of 200 ppm in their kidney it would take 352 μg of Cd/day for 50 years. In the derivation of the cadmium drinking water standard, EPA assumed a daily dietary exposure of 75 μg and 20 μg Cd/day from water. This 20 μg Cd/day from drinking water would result from a standard of 10 μg/L. The total daily Cd exposure is therefore approximately 95 μg Cd/day and thus a safety factor of 4 was assumed.

In proposing their drinking water standard for cadmium, EPA requested feedback from the public as to whether the standard should include additional protection for cigarette smokers, since smoking is a source of appreciable cadmium exposure (e.g. approximately 1.5 μg Cd/cigarette) [4]. It is interesting to note that of the 52 comments received by EPA on this issue, only three suggested that this standard be modified to include protection for the cigarette smokers. The EPA administration agreed with the majority of public responses [11].

RECREATIONAL WATER AS A COMMUNITY DRINKING WATER SOURCE

Of particular relevance to this conference is a situation where recreational water such as the Connecticut River has been repeatedly proposed for use as a drinking water source for Metropolitan Boston some 90 miles away via a divergence project. If this does come to pass, then the water quality of the Connecticut River must be consistent with EPA drinking water regulations. Several questions may arise such as is the Connecticut River water capable in itself of meeting EPA standards or if it can't, then would it be sufficient if the Connecticut River water plus the cleaner water to which it is added (i.e. Quabbin Reservoir water) meets EPA regulations? In anticipation of this potential situation, it is important to realize that the Connecticut River is adjacent to Interstate Highway Rt. 91 which is heavily salted. Since inadvertant contamination of numerous drinking water sources via deicing practices with NaCl is a major problem in Massachusetts, the possibility that the Connecticut River may have high levels of sodium contamination was apparent. Since the State of Massachusetts has a sodium standard of 20 mg/L which is designed to offer protection to those with cardiovascular disease, it was important to evaluate seasonal variation in sodium levels of the Connecticut River. We have recently completed such a study and have determined that the sodium levels are not of health concern, being under 10 mg/L. However, the possibility that recreational waters may be used for drinking water is real and their potential effects on high risk groups is a concern.

STANDARDS FOR CARCINOGENS: ROLE FOR HIGH RISK GROUPS?

There is considerable uncertainty concerning approaches to setting standards for carcinogens. In general, everyone has been considered susceptible to carcinogenic agents. Consequently, groups at high risk to environmental carcinogens, with the obvious exceptions of smoking behavior as a predisposing factor to asbestos, uranium and coke oven emission related cancer, have not generally been emphasized. Most approaches have been to reduce the exposure to the lowest practical or detectable levels (e.g. vinyl chloride) or to attempt to find suitable replacement materials. However, evidence is emerging from developmental, genetic, and nutritional areas that there are specific and identifiable biological factors which may greatly enhance one's susceptibility to develop an environmental related cancer [4,6].

People with the inherited disease, xeroderma pigmentosum, which is characterized by a significantly reduced capacity to repair damage to DNA caused by ultraviolet light, are known to be at exceptionally high risk to UV induced skin cancer [12,13].

Other studies have suggested that individuals with a high capacity to induce the synthesis of the enzyme, aryl hydrocarbon hydroxylase in lung tissue may be at considerable risk to developing cigarette smoking related bronchogenic carcinoma [14,15]. Lower [16] has recently presented a strong case based on animal models and limited human epidemiological data that individuals who are slow acetylators of arylamines may be at increased risk of bladder cancer following exposure to these dyes. There is also a voluminous literature which clearly has collectively demonstrated an inverse relationship between levels of vitamin A in the diet and susceptibility to hydrocarbon induced epithelial cancers [17,18]. In more practical terms, the enhanced susceptibility of the unborn and young children to x-ray induced cancer has been long known and special precautions have been made to reduce x-ray exposure to these groups. A careful examination of these and other supportive studies leads to the conclusion that all individuals are not equally susceptible to experience environmentally-induced carcinogenesis and in many cases the differential susceptibility may be orders of magnitude [4]. Thus, biological factors may markedly influence each individual's own threshold response to numerous environmental carcinogens.

When regulatory agencies develop risk assessments for carcinogenic agents, it is necessary that they differentiate between the "general population" and those considered at high risk. Knowledge of both the identification and quantification of those at increased risk provides decision makers with a theoretical framework from which to derive their cost/benefit assessments. An example of how the knowledge of high risk groups may be utilized in the assessment of risk following exposure to environmental carcinogens can be seen in the work of Gofman and Tamplin [19]. They attempted to predict the incidence of cancer per 10^6 persons per year due to various levels of radium^{226} in the drinking water with the intention of defining an acceptable level of exposure. Of relevance here is that Gofman and Tamplin assumed that the latency period is shorter for in utero exposure than otherwise (i.e. 5 years vs. 15 years for all radiation exposure beyond birth). This special consideration of the increased susceptibility of the unborn to radiation induced cancer resulted in greater estimations of cancer risk than other methodological approaches (i.e. Evans and EPA) which utilized the traditional extrapolation process of predicting carcinogen effects at low doses based on data at high levels of exposure in adults [19].

At the present time, EPA's carcinogen policy assumes that there is no scientific basis for estimating "safe" levels for carcinogen exposure. This is based on the recognition of methodological limitations which do not permit the establishment of a threshold for carcinogenic effects. EPA has evaluated the availability of linear biostatistical models (such as the one-hit, logit, probit, Multi-Stage, etc.) to estimate cancer risks at various exposure levels greater than zero. The major point regardless of the model is that there is no safe level regardless of how low the level may be.

How do these models relate to and encorporate the concept of increased susceptibility? The dosage effect relationships of these respective biostatistical models are primarily descriptive of the variation in susceptibility between the individuals of a population. Thus, knowledge of increased susceptibility is consistent with the premises of the linear models since it accepts linearity with respect to how a genetically and culturally heterogeneous population may respond to environmental exposures, and in fact, provides the biological basis to interpret such dose-effect relationships.

The models assume that the conditions of increased suscepbility are randomly distributed throughout the population, even though the actual condition affecting susceptibility is probably deterministic. The models predict the effects of low level exposure in a population which is similar to the population used in the original high dose study. Since most animal studies use highly inbred strains, identical diets and living conditions, the extrapolation is valid only for similar individuals under the identical living conditions. Consequently, if the original study did not take into account factors which result in the high risk condition, then the downward extrapolation would not offer the same degree of protection for potential high risk groups. The question arises as to whether the model should be applied to a high risk group. For example, biostatistical models have been used to predict the occurrence of aflatoxin-induced cancer in humans from consumption of peanut butter sandwiches with trace amount of contamination based on studies with normal rats. However, since dietary inadequacies are known to enhance aflatoxin induced cancer, should the biostatistical model be applied to the high risk condition? Thus, to answer the original question posed above, the linear models are not attempting via a generalized methodology to predict the occurrence of environmentally induced cancer amongst the various high risk segments of the population unless they were included in the original study from which the data will be extrapolated.

CONCLUSIONS AND RECOMMENDATIONS

The EPA has attempted to consider the special sensitivities of those at increased risk to pollutants in the standard setting process for air and water pollutants (Table 1). It is interesting to note that EPA has specificially considered both developmental and pre-existing disease conditions, that is, the increased susceptibility of the very young to nitrate related formation of methemoglobinemia and the increased sensitivity of children to certain aspects of lead toxicity [20], and the increased risks of those with various cardiovascular diseases to respiratory tract irritants. Despite the fact that EPA has made a good faith effort to incorporate protection for high risk groups in the setting of certain standards, a more specific and consistent policy toward the utilization of the knowledge of high risk groups in environmental health policy in general and standard setting in particular is needed. Consequently, the following recommendations are made to assist in the development of such a policy orientation.

1. A strong and consistent philosophy should be maintained which is designed to insure the health of all members of society including those at increased risk, to the extent feasible.

2. Contrary to many popular notions, high risk groups do not always comprise a very small, numerically insignificant, portion of the population. In fact, with regard to those pollutants such as nitrate which have a profoundly differential affect on the organism depending on developmental factors, all individuals are at increased risk for a certain period of time during their ontogeny. Even with regard to certain genetic disorders which affect only a very limited segment of the total population, a disorder such as an erythrocyte G-6-PD deficiency affects approximately 13% of the American Black male population. In a city like Chicago, this would amount to nearly 100,000 affected males, not an insignificant number.

3. The concept of multiple predisposing conditions superimposed on individuals or groups resulting in an exaggerated high risk condition should be further developed. For example, an individual having both a G-6-PD deficiency and medically low levels of vitamin E in blood sera, would be considered as a very high risk group to oxidant stressor agents. The estimated number of this possible very high risk group in Chicago alone would be approximately 6,000 [4].

4. No serious efforts have been made to deal with how knowledge of nutritional factors which modify pollutant toxicity should be dealt within a standard setting framework. Recently EPA has begun to try to deal with this complex issue. In proposing the national ambient

TABLE 1. HIGH RISK GROUPS IN THE DERIVATION OF U.S. ENVIRONMENTAL HEALTH STANDARDS

a. Drinking Water Standards

Substance	High Risk Condition Considered
Arsenic	none
Barium	no specific groups; but a safety factor of two incorporated to account for variation (or increased susceptibility) within the human population.
Cadmium	none
Fluoride	children-prevent mottling of teeth
Lead	children-prevent neurological disorders
Mercury	those humans who exhibited toxicity at the lowest levels of exposure from a group of mercury poisoned adults.
Nitrate	infants-protect against methemoglobinemia
Selenium	none
Sodium (no standard)	those with heart and kidney diseases
Chlorinated Hydrocarbon Insecticides (non-carcinogenic)	none

b. Air Standards

Carbon monoxide	heart disease
Hydrocarbons	none
Lead	children-prevent blood disorders
Nitrogen Dioxide	young children-respiratory infections
Ozone	asthmatics
Particulates	aged, those with cardiovascular disease
Sulfur dioxide	aged, those with cardiovascular disease

air quality standard for lead, EPA specifically cited the enhanced susceptibility of certain individuals with inadequate diets [20] to lead toxicity. However, they did not specifically derive their standard for lead based on dietary factors predisposing lead toxicity since it was felt that there was insufficient data to determine the effects threshold for such groups or to accurately characterize such groups in the population. This is clearly an emerging area in which greater knowledge could profoundly affect future standards.

5. Knowledge of high risk groups in terms of both their identification and quantification should be used in the development of cost/benefit analyses.

6. An area which has generally been avoided until recently by EPA in setting standards has been the issue of carcinogens. The first ambient air standard for a carcinogen (i.e. benzo(a)pyrene(BAP) was promulgated in the Soviet Union [6]. It was designed to prevent the occurrence of cancer in the normal lifespan of the average individual. It assumed that as the exposure level decreased the length of the latent period increased. Thus, even assuming a non-threshold dose-response relationship, BAP-induced cancers would theoretically occur after death by other causes. Therefore, it was of no practical consequence. This approach, if verified by future studies, could be supplemented by incorporation of data from segments of the population who may be at increased risk to developing pollutant induced cancer.

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HUMAN HEALTH EFFECTS IN DEVELOPMENT OF CRITERIA AND STANDARDS

J. Kennerly Davis, Jr.

Hunton & Williams
707 East Main Street
Richmond, VA 23212

INTRODUCTION

EPA's health criteria, those proposed in 1979 as well as those being developed for the future, raise several issues of great importance to national water quality policy. One central issue concerns the adequacy of the data bases that have been used to formulate many criteria. Another important issue concerns the soundness of the methodologies that have been used by the Agency to derive their proposed criteria.

These water quality issues arise, of course, within the larger context of the Clean Water Act as a whole. Therefore, an evaluation of the Agency's criteria proposals should begin with an overview of the sections of the Act relating to water quality.

OVERVIEW OF THE CLEAN WATER ACT

The Clean Water Act provides for a number of interrelated water pollution control programs. Separate roles have been carefully defined by the Act for the federal government and state governments to carry out these programs. According to Section 101(b) of the Act, the states have the "primary responsibilities and rights" in the overall plan. One of the principal pollution control programs contained in the Act is based on the formulation and implementation of water quality standards, which define the allowed concentration levels for specified pollutants in particular waters. States have had the responsibility for setting water quality standards since 1965, and the passage of the Water Quality Act.

Under Section 303(c)(1) of the Act, states are to review and "as appropriate," modify their water quality standards at least every three years. Section 303(c)(2) of the Act directs states to set standards consisting of the designated uses of the navigable waters involved and the water quality criteria for such waters to support those uses. The states are to set standards for their waters only after taking into consideration a wide range of possible uses for those waters, as well as the value that would accrue from one use or another. Possible uses include public water supply, fishing, swimming and agriculture. It may seem an obvious point, but it should be noted that each of these uses does indeed provide for the use of the water in question. Congress never intended that water quality standards be used to prohibit the use of waterbodies for any purpose.

To meet these statutory requirements and to fulfill their role in the formulation of water quality policies, states must have freedom to make the necessary decisions as to use designations and local water quality criteria. States must control use designations because they require policy judgments about the allocation of scarce local resources that only the states can properly make. States must control their criteria because criteria must be appropriate to local conditions.

Under the Act, the federal government also plays a role in the formulation of water quality standards, but it is complementary, and in many ways subordinate, to that of the states. As a first step, Section 304(a) of the Act calls on EPA to formulate national criteria for water quality to guide the states in setting their own criteria under Section 303. If properly developed, these national criteria can provide significant technical assistance without interfering with the state freedoms so essential to the overall success of the water quality standards program.

EPA CRITERIA

Section 304(a)(1) of the Act requires EPA to develop and publish criteria for water quality that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare, including fish, which may be expected from the presence of pollutants in water.

The nature of these criteria is informational. They are not in themselves enforceable. Criteria are issued to the states and the public under Section 304(a)(3) to assist the states in the formulation of water quality standards. The rationale is to draw upon the national resources and expertise of the Agency in an effort to ensure that the state standards ultimately derived are founded on valid scientific information and are otherwise in accordance with the Act.

EPA explicitly recognizes this purpose of informational assistance in its July, 1978 Advance Notice of Proposed Rulemaking on water quality standards policy. 43 Fed. Reg. 29588-89. In its March, 1979 notice on water quality criteria, EPA stated that publication of such criteria has "no regulatory impact on any party" until enforceable water quality standards are set by administrative rulemaking procedures. 44 Fed. Reg. 15926.

Despite these statements, EPA has indicated that it is considering whether to assert what would amount to presumptive federal control over all aspects of water quality standards policy. 43 Fed. Reg. 29588. Under this new EPA policy, the pollutants to be covered by water quality standards would be designated by EPA; designated waterbody uses would as a general matter be limited to "fishable, swimmable" status, and the criteria supporting the designated uses would be national criteria established by EPA, unless a state was able to establish that those criteria were inappropriate. This paper later discusses the ways in which such policy pronouncements are contrary to the Act and do not assist sound environmental decision making.

Scientific Validity of EPA Criteria

Section 304(a)(1) of the Act requires that EPA develop and publish criteria "accurately reflecting the latest scientific knowledge" about water quality. This ambitious standard necessitates that criteria posses certain technical features. At a minimum, they should be derived from comprehensive and sound data bases, using methodologies that are able to take account of the many factors that influence the ultimate impact of a substance. There are several features which must be present in scientifically valid criteria.

Data bases for criteria should be assembled to include all studies which were conducted and reviewed in accordance with generally accepted, sound scientific procedures. These data bases should be adjusted and updated to incorporate the sound elements of subsequent studies. The quality of the studies that are included in data bases should be characterized by the use of confidence limits. The Agency should also characterize the criteria derived from these data by the use of two or three broad classifications or categories to express different levels of Agency confidence in the criteria. Data characterization and criteria would enable others to assess the scientific validity of EPA's conclusions. National criteria can never be taken to be presumptively applicable in the formulation of state water quality standards. Their usefulness to the states depends, in the first instance, on their scientific soundness.

Health criteria should be derived to identify and regulate only the forms and valence states of pollutants that are relevant to the protection of human health. EPA has begun to do this, and deserved credit for taking such steps. Also, they should be derived in such a way as to account for the significant localized environmental factors such as salinity and hardness that can influence the effect of a chemical substance on human health. If data on a certain influencing factor are lacking, or insufficient to be usefully employed, this should be explicitly recognized and should influence the confidence estimates described above.

Health criteria should be derived to take account of the observed benign and beneficial effects of substances. Many substances for which criteria are derived occur naturally; a number of criteria substances are essential to human health. Criteria should be derived in a way that recognizes the important differences in response of different population subgroups. The impact of a substance on human health may vary significantly according to the sensitivity of the particular population subgroup in question. Health criteria should also be formulated in a way that permits coordination with other health-related programs such as those under the Safe Drinking Water Act (42 U.S.C. §§300f-300j-10) (1976), The Occupational Safety and Health Act, (29 U.S.C. §§651-678) (1976), and the Pure Food and Drug Act (21 U.S.C. §§301-392) (1976). Only if the special regulatory emphasis of each program is taken into account and coordinated can the total health benefits to society be maximized. For instance, though one possible use of water is for drinking water supplies, there is no need to achieve end of pipe consumption quality with Section 303 water quality standards. Significant treatment is applied to water after it is taken from its source of supply and before it is consumed.

Regulatory coordination is especially necessary if criteria documents are drawn up taking account of the amounts of a substance that a person takes in through exposure routes other than water. If the water route contributes most of the dose of a substance, then one regulatory response may be called for. If the water route contributes only an insignificant amount of the dose, then quite a different regulatory response may be called for. Relative exposure route significance is best considered at the stage of the process when state water quality standards are set. The most appropriate and efficient regulatory response can be selected only if due regard is given to factors of economics and technological feasibility.

Criteria are scientific entities. They must be objective and neutral. Other sections of the Clean Water Act provide for the consideration of non-scientific, policy-based factors. States, for example, consider social and economic factors when they set water quality standards under Section 303. However, such factors play no role in the formulation of national criteria by EPA. When questions arise under Section 304 programs about the adequacy of data or the selection of a model, these questions must be answered with the best available scientific judgments and not policy judgments that may be unrelated or even contrary to accepted scientific procedures. It may or may not be appropriate to include "safety factors" in the pollutant concentration limitations that are ultimately established for a waterbody. But these non-scientific factors should be addressed explicitly and, if used, should be incorporated at the standards-setting stage of the regulatory process.

Finally, when developing criteria, the Agency must consider whether precise and accurate, and fully validated analytical methods were used to conduct the studies in the criteria data bases. The Agency must also consider whether such analytical methods are currently available for use by the states in their water quality standards programs. The scientific validity of criteria can only be assessed by currently available analytical techniques. Moreover, the practical effectiveness of any water quality standards program depends on technically valid monitoring and enforcement techniques that are based on analytical equipment that is suitable for field use, generally available and not unreasonably expensive to operate.

Statutory Consistency of Criteria Implementation

The basic purpose of water quality criteria is to provide technical information to states to assist them in setting water quality standards. The criteria derived by EPA should be developed in a way that assists states in making use designations and in selecting the local, most appropriate criteria to support those uses. As discussed above, EPA's criteria must be scientifically sound to fulfill this role. However, they must also be formulated and issued in a manner that is consistent with and supportive of the primary role of the states in setting and implementing water quality standards. The Agency can accomplish this by applying the logic of risk assessment analysis to the development of water quality criteria.

In its guidelines and methodology used to prepare human health criteria, EPA recognizes the critically important fact that the risks posed by ambient water concentrations of pollutants cannot be entirely eliminated. 44 Fed. Reg. 15926, 15974-75 (March, 1979). Unfortunately, EPA does not follow through on its important insight concerning risk. In developing water quality criteria for the protection of human health, the approach used by the Agency for carcinogens involves only very limited risk assessment analysis; the approach for non-carcinogens involves none at all. The Agency should act on the implications of its recognition of risk and greatly expand the use of risk assessment analysis in the formulation of water quality criteria. Such analysis is relevant for the formulation of all criteria, aquatic as well as health.

Risk assessment is a two-stage process. The first stage involves the identification and measurement of the risks associated with the subject matter of analysis. The second stage involves the social evaluation of the risks that have been identified, and the selection of an appropriate response to those risks. The two stages of this process correspond logically to the two stages of setting water quality standards under the Clean Water Act, with the science-based task of risk measurement assigned to the federal government and the policy-based risk evaluation assigned to the state government.

Criterion formulation by the federal government is, by its nature, an exercise in risk measurement. The activity is scientific. The factual descriptions presented must be objective and accurate. On the other hand, the setting of standards is by its nature an exercise in risk evaluation. The activity is policy oriented. The designation of uses and local criteria involves the consideration of social, economic and political factors to determine the appropriate public response to the risks that have been measured.

To aid the states in making use designations and in adjusting those designations to local needs, aquatic as well as health criteria should be issued to the states in the form of concentration-response curves or their equivalent for each substance. If criteria are issued as concentration-response curves which reflect the full range of effects from the substances of concern, states will be able to make intelligently the use designations required under the Act. National criteria should provide the states with a package of information useful in making decisions about the appropriate concentration limitations for a substance. They must not be used to impose the federal government's own decisions about concentration limitations on the states, contrary to the logic of risk assessment analysis and the provisions of the Clean Water Act.

Aquatic as well as health criteria should be derived by EPA in a way that will also aid in the selection of state criteria. To accomplish this, EPA should publish for each substance sets or "families" of concentration-response curves for each substance that recognize and allow adjustment for the significant factors which can influence the ultimate effect.

Issuing criteria as concentration-response curves is entirely consistent with the requirements of the consent decree in *NRDC v. Train*, 8 ERC 2120 (D.D.C. 1976). The consent decree calls for criteria "accurately reflecting the latest scientific knowledge on the kind and extent of all identifiable effects on aquatic organisms and human health of each of the [65] pollutants listed in Appendix A." 8 ERC 2127. The consent decree also says that these "criteria shall state, *inter alia*, for each of the pollutants . . . the recommended maximum permissible concentrations (including where appropriate zero) consistent with the protection of aquatic organisms, human health, and recreational activities." *Id.* Concentration-response curves are needed to adequately and accurately reflect "all identifiable effects" of pollutants in waterbodies. Such curves are also required to indicate the maximum permissible concentrations called for. The maximum concentrations with the protection of aquatic life or human health are not constant for each pollutant; they will vary from waterbody to waterbody and case to case depending on the interaction of chemical environmental and other factors.

THE UTILITY OF WATER QUALITY CRITERIA AND STANDARDS FOR PROTECTING AQUATIC BIOTA AND AQUATIC ECOSYSTEMS

Peter V. Hodson

Great Lakes Biolimnology Laobratory
Canada Centre for Inland Waters
Burlington, Ontario, Canada

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ABSTRACT

Water Quality Criteria and Standards attempt to limit impact of toxic substances on aquatic biota by establishment of "safe" concentrations in water, or other ecosystem compartments, based on a review of cause-and-effect relationships observed in laboratory aquatic bioassays. Such criteria provide perspective for decisions on water use and have an immense value as regulatory tools, goals for water quality management, targets for waste treatment technology and scales against which to compare results of monitoring.

Implicit in water quality criteria are a number of technical weaknesses including gaps in the data base required to set criteria, failure to maintain a consistent philosophy when setting a large number of criteria, and a failure to resolve problems in application (e.g. mixing zone allocation).

The most important and fundamental problem, however, is that the water quality criteria approach has a number of deficiencies as a management strategy, and it cannot be relied on as the only strategy for protecting aquatic ecosystems. Problems include failure to examine the systems to be protected, to adequately limit impact of mixtures of toxicants and environmental interactions with toxicity, and to minimize loading.

Clearly additional complementary strategies are required such as ecosystem health assessment, ecosystem quality criteria, expanded water quality criteria (including loading limitations) and end-of-pipe standards based on whole effluent toxicity bioassays.

INTRODUCTION

Water quality criteria were first well-documented by McKee and Wolf (1963). There have since been a variety of state, provincial and federal government publications of criteria to protect a variety of water uses ranging from health of aquatic organisms to industrial processes. These criteria have reflected a rapidly expanding data base, a variety of philosophies and procedures for recommending safe limits, an expanded list of chemicals considered, and criteria that are successively lower in each new document.

The water quality criteria approach has been widely adopted because of its obvious utility as a scientific basis for standards, guidelines and regulations. In addition, criteria provide goals for water quality management and waste treatment technology, they are a scale against which to compare results of monitoring, and they provide a perspective in decisions on water use. Despite these advantages, criteria are not an end in themselves, but a means to an end: the protection of specified beneficial uses of water. In the case of aquatic ecosystems, protection will not be achieved if: (1) we fail to recognize major weaknesses in the water quality criteria approach; and (2) this approach is our only method of protecting aquatic ecosystems.

I would like to emphasize that this critique is not intended to reject the water quality criteria approach, but rather to illustrate its weaknesses and to suggest complementary management and research strategies that are necessary for protection of aquatic ecosystems.

DISCUSSION

Some of the weaknesses of the water quality criteria approach have been reviewed in a report of the International Joint Commission (IJC 1975). Unfortunately, the thoughtful discussions and caveats of this report are rarely considered or understood by those adopting or using criteria. The weaknesses may be roughly divided into three areas:

1) Deficiencies in development of criteria;

2) Deficiencies of criteria as a management tool;

3) Deficiencies of the criteria approach as a management strategy.

1. DEFICIENCIES IN THE DEVELOPMENT OF CRITERIA will result in recommended limitations that are poorly founded scientifically or are inappropriate to protect a desired use. Examples of problems include:

a) Components not protected

The majority of criteria are derived from laboratory studies with fish, zooplankton.and algae. Consequently, classes of organisms in other ecological niches may not necessarily be protected by these criteria. For example, benthos cannot be assumed to react solely to waterborne contaminants, as may be the case with a free-swimming bioassay fish. Instead, they may react more strongly to contaminants in interstitial waters, in ingested sediment or adsorbed to organic particulates. Similarly, wildlife and aquatic birds may acquire contaminants from diets centred heavily on aquatic biota. Criteria limiting contaminants in fish flesh to protect human health (e.g. PCBs) may not adequately consider the high daily dose ingested by wildlife and their unique dose-response relationships. Clearly, if we wish to protect these organisms as essential components of aquatic ecosystems, the data base must be expanded.

b) No Ecosystem-based criteria

Bioassays of single species ignore the complex and often subtle interrelationships between predators, prey, parasites, competitors, etc. Slight alterations in these interrelationships cannot be measured in these bioassays and could cause significant changes in ecosystem structure and function.

Therefore, protection of the individual species from direct contaminant effects does not necessarily protect the ecosystem.

c) Inappropriate criteria

Failure to interpret bioassay data correctly, to appreciate the significance of observed responses to contaminants and to understand interactions between toxicity and water quality leads to inappropriate criteria. Good examples of this problem are recommendations of unique criteria for each of several water hardness ranges, based on observed variation of toxicity with hardness (EPA, 1976). Unique criteria for each of several hardness ranges implies that toxicity changes in a step function at an arbitrary boundary between ranges of hardness. Since the toxicity-hardness relationship is really a continuous function, the criteria fail to protect aquatic biota in waters of fluctuating hardness or hardness close to the boundaries of a range.

2. DEFICIENCIES AS A MANAGEMENT TOOL result from a failure to fully develop the philosophy and procedures required to apply recommended criteria to real situations. There are two outstanding examples:

a) Mixing zones

Mixing zones are those portions of a water body in which effluents are mixed and assimilated by the receiving water. Criteria are generally meant to be met at the edge of the mixing zone so that the area within is a recognized zone of degradation and represents a loss of beneficial uses. Mixing zone guidelines have been published that limit degradation to prevent objectionable colour, odour, deposits, floating materials, conditions that are rapidly lethal to aquatic biota, bioconcentration of persistent contaminants, blocking of zones of passage, etc. (IJC, 1975). Since 1975, more than 30 specific water quality objectives or criteria have been written and formally adopted in an international treaty between Canada and the United States (Canada-U.S. Agreement on the Great Lakes, 1978) but, to my knowledge, no one has allocated a mixing zone in the Great Lakes system according to the ideas originally conceived. The concepts and procedures for mixing zone allocation have not advanced significantly in pace with criteria. The result is that criteria cannot be used directly to limit discharges or as a basis for designing treatment systems.

b) Definition of a violation

There are few guidelines for what constitutes a violation of an objective. A statistical requirement for chemical surveillance data may be established (e.g., significantly different as determined by a t-test at a given level of significance), but what does it mean ecologically? Can an aquatic ecosystem tolerate a 1, 10 or 100% deviation from the objective? For how long? How frequently? What temporal and spatial sampling grid is required? When a violation occurs, what is the appropriate action?

Therefore, it is clear that the water quality criteria approach is not a fully-developed management tool and we do not have adequate procedures for applying it in a meaningful way to protect aquatic ecosystems.

3. DEFICIENCIES AS A MANAGEMENT STRATEGY of the water quality criteria approach represent the most serious set of problems because it does not deal directly with ecosystem health or loading.

a) Ecosystem health

Parts per million is not a meaningful measure of ecosystem health. Reliance solely on chemical criteria and monitoring of water quality on a "pass-fail" basis will protect ecosystem health only by coincidence or good luck. The situation is analagous to human health. All the requirements for health, such as clean air and water, good food, social stability, etc., can be met according to numerical guidelines. However, there is no guarantee that the population is not affected by some unproscribed factor, or an unusual combination of seemingly innocuous factors. Only by assessing community health through personal medical examinations and epidemologic surveys can it be stated that the population is healthy. Similarly water quality criteria are only an aid to controlling ecosystem health. They can limit the occurrence of known stressors or contaminants, for which adequate data exist to set a numerical limitation, but they cannot limit undetected or unknown substances, or unexpected responses to known substances.

There are two good examples of this problem:

(i) Mixtures--

The huge potential for multiple toxicant interactions is often recognized. However, we are presently unable to formulate specific criteria to prevent mixture toxicity due to the vast number of permutations and combinations of contaminants. Recent research demonstrating significant algal toxicity of heavy metals mixed together at the concentrations recommended as safe by IJC objectives clearly illustrates the potential problem (Wong et al., 1978).

Environmental interactions present a similar problem. Toxicity of a contaminant will vary with acidity, hardness, temperature, etc. Although we recognize the potential for interactions, resources are inadequate to research all possible combinations of contaminants and environmental variables.

(ii) Unusual dose-response relationships--

"Specific water quality objectives were designed to be met at the periphery of mixing zones. This assumes water quality conditions better than the objectives will result beyond the mixing zones" (IJC, 1975).

The objectives and mixing zone concept assume a situation in which biota within a mixing zone receive a greater exposure (concentration X time) than those outside a mixing zone. Those within the zone, while protected from acute mortality, may exhibit adverse responses, while those outside the zone are theoretically protected. Unfortunately the mixing zone is not a magic barrier to movement of aquatic biota. Consequently, through passive or active migration, aquatic organisms encounter toxicants at concentrations typical of mixing zones. Despite short transit times detrimental effects may occur. For example, short-term exposure of minnows to copper when sexually mature has been shown to have the same effect as complete life-cycle exposures (Pickering et al., 1977). In Placentia Bay, Newfoundland, herring migrated through a mixing zone containing colloidal elemental phosphorus. Weeks later, at great distances from the mixing zone, they died of hemorrhaging (Jangaard, 1972). The fish were obviously protected from short-term mortality but not from the unusual cumulative or delayed effects of one short-term exposure.

b) Loading

Contaminants in the ecosystem may be divided into two classes: conservative and non-conservative substances. Conservative substances are those that do not degrade, or degrade very slowly, and include the elements (metals, chloride, etc.) and organics such as mirex, PCBs, HCB, etc. Non-conservative materials are those that degrade at a measurable rate and include most organic molecules.

In a mixing zone, concentrations of toxic materials should be reduced to the criteria levels through dilution, precipitation, and degradation. For non-conservative materials, these processes may be complete within the mixing zone. However, conservative materials may only be diluted to background levels in water or sediment, given sufficient volume. In extreme cases, background concentrations on a lake-wide basis might rise to equal the concentration in the mixing zone. For example, mercury is non-conservative in water but, after precipitation, it appears relatively conservative in sediments. Since the sediments and particulate matter of the Great Lakes move considerably with currents, large portions of the lake bottom "down-stream" of contamination are now sources of methyl mercury to local food chains (IJC, 1977). Therefore, while water quality criteria may have been met locally, mercury in sediments moved out of the mixing zones and into the remaining parts of the lakes.

To limit the rate of mercury contamination of fish, we must obviously limit the rate of accumulation in the sediments. For conservative materials like mercury this means limiting lake-wide inputs on a total loading basis that considers both point-sources effluents and diffuse sources, such as atmospheric inputs, rivers, shoreline erosion, etc. For some contaminants, the requirement may be zero input.

Non-conservative substances must also be treated the same way since the "assimilative capacity" of an ecosystem is limited. The major difficulty with this approach is that "assimilative capacity" is usually defined in broad engineering terms (e.g. oxygen sag curves) rather than in terms of ecosystem structure and function. The effect of contaminants on the "assimilative capacity", and the interactions between organic enrichment and toxicity and bioconcentration of contaminants, have yet to be fully defined.

c) The cost of changing criteria

Water quality criteria are traditionally revised as new data becomes available and each revision produces criteria that are more stringent than those advocated previously. While no one would argue that the criteria should be refined as our knowledge increases, industrial compliance to each revision requires either a larger mixing zone (with potential adverse environmental effects) or additions to, or reconstruction of, existing waste treatment systems. Obviously, if the waste treatment capacity was "over designed" at the start, or if a policy of zero effluent discharge were adopted, changes to criteria would be less important.

COMPLEMENTARY STRATEGIES

The discussion so far has illustrated a variety of problems that limit the utility of the water quality criteria approach. Potential solutions to these problems include:

1. EXPANDED CRITERIA

Future criteria should be based on a wider perspective that includes:

a) Concentration criteria for more ecosystem components including water, sediment, particulates and biota to protect all classes of aquatic organisms;

b) Concentration criteria based on studies of contaminant effects on ecosystem structure and function;

c) Concentration criteria that correctly interpret toxicity interactions with water quality;

d) Loading criteria based on an understanding of fate and effect of contaminants, recognizing the conservative nature of many contaminants and the limitations of "assimilative capacity" of aquatic ecosystems; and

e) Ecosystem criteria that define the structure and function of the desired state of a water body satisfying defined beneficial uses and that identify key organisms whose presence, successful growth, and reproduction are indicative of that desired state.

2. MIXING ZONES

Research should be directed towards practical mechanisms for mixing zone allocation that ensures a desired level of protection for aquatic ecosystems and a practical means for planning waste treatment systems.

3. ECOSYSTEM HEALTH ASSESSMENT

Ecosystem health assessment would indicate whether water quality criteria and other vaste management strategies were protecting the desired use. Elements would include:

a) Assessment of ecosystem structure and function to ensure it supports and represents the desired beneficial uses;

b) Intensive study of ecosystems within and immediately adjacent to mixing zones to provide early warnings of adverse effects; and

c) Assessment of fish health to detect physiological responses of fish as an early warning of adverse effects.

4. EFFLUENT CRITERIA

The study of real effluent mixtures, and interactions of effluent toxicity with real environmental variables will be more efficient than studying individual interactions in the laboratory. The result will be criteria for specific point source effluents that will limit site-specific impact. Studies of chronic effluent toxicity will also provide useful responses that can be used as monitoring tools to define criteria violations.

In summary, it is clear that there are many inadequacies in the water quality criteria approach to protection of aquatic ecosystems. Many of these can be solved through improvements to the scientific data base and the way the data base is used. The remainder could be solved through application of complementary strategies to control and assess impact of contaminants. Complementary strategies would be most successful if they were applied in a coordinated fashion by one responsible group following a comprehensive plan. Obviously, this requires initiative from the top!

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STATE RESPONSE TO FEDERAL PROPOSALS FOR WATER QUALITY CRITERIA FOR TOXIC SUBSTANCES

C. S. Olson, J. Park, and D. J. Schaeffer

Illinois EPA
2200 Churchill Road
Springfield, IL 62706

ABSTRACT

The response of a state to a federal proposal for a water quality standard for a substance might take the following form: preliminary assessment of whether the substance is present in the waters of the state; gathering of chemical, analytical and toxicological information on the substance; assessment of applicability of information to waters in that state; gathering of information on the treatability of the substance; integration of all information to determine whether the criterion is enforceable as a standard. This process is discussed in terms of Illinois' experience with cyanide and our preceptions of the problems facing the state as it tries to develop standards for the priority pollutants. Decisions on what substances to regulate, qualifications required of personnel making the evaluations, and the standards setting process are discussed. Portions of Illinois' toxics program relevant to these processes are described.

INTRODUCTION

States already have extensive experience in dealing with water quality standards for a number of substances on the priority pollutant list, and TSCA may control the environmental impact of "certain" additional substances. The evaluation of the hazard posed by substances on the priority pollutant list is an effort by USEPA to initiate the control of these many compounds. Beyond these compounds there are many other potentially hazardous substances which may have to be regulated in the future. Thus, we estimate that 2500-9000 compounds are distinguishable by present gas-chromatographic-mass spectrophometric (gc/ms) analyses with extraordinary effort (Janardan and Schaeffer, 1979b, 1980a, b; Schaeffer and Janardan, 1980); these represent only a fraction of the 18,000-28,000 compounds observable as peaks on gc/ms. All of these are potential candidates for inclusion on a "priority pollutant" list under the present regulatory philosophy. USEPA is presently re-analyzing all of its computerized gc/ms analytical tapes in an effort to identify as many substances on these as possible. Their intent is to ascertain whether there are other substances present in the samples which should be added to the list. We do not know the status of this work.

We believe the priority list was poorly chosen and that we can develop a list of substances more relevant to Illinois. For instance, we have recently reported that of over 200 different compounds identified in 16 samples of industrial effluents and river water, 74 are carcinogens or promoters (Somani et al. 1980). Some of these potentially hazardous substances are on the priority pollutant list while others are not. It is evident that as more and more compounds are proposed for regulation, there will be many of only local or regional interest. Therefore, one of our chief concerns is that states be given leeway in deciding which substances to regulate instead of being forced to set standards for all.

The large volume of technically complex work required to support an increased standards-setting and monitoring effort suggests that states should hire people with expertise in aquatic toxicology, mammalian toxicology, cytogenetics, analytical chemistry of trace organic compounds, etc. While some existing staff can be trained to perform these functions, in-house training for such activities as mutagenicity testing is infeasible. Because these types of skills are already in such high demand, people having them command premium prices for their services (Weir, 1980); but, most state civil service regulations prohibit paying salaries in the required range of $30-60,000. Thus, most states will probably have to rely on outside groups such as regional commissions, the American Fisheries Society, and ASTM for information, unless they elect to accept USEPA proposals uncritically. For toxicology testing and monitoring they may have to rely on private consulting organizations. This may present a conflict since these consultants are also working for the industries being regulated. In our experience over the past three years, the remaining alternative of training existing staff is really a non-alternative because of salaries and job mobility. (All of the people we had working on our toxics monitoring easily secured much better paying jobs elsewhere.) Nevertheless, we think that state government is where the "buck stops," and that in each state there should be a small group responsible for understanding what is involved in establishing criteria and standards for substances whose effects on the environment are largely unknown and for which the costs of regulating are, and probably will remain, several orders of magnitude higher than those of the pollutants which have traditionally been regulated.

Another general concern is how a large number of compounds can be handled by a lengthy standard-setting process. Illinois' law requires the Illinois Environmental Protection Agency (IEPA) to make a proposal for a standard. This undergoes review by three independent bodies, two sets of public hearings and an extensive, independent economic impact study must be completed before new regulations are adopted. There are required waiting periods at different points, and the whole process takes from 1 to 3 years, or more. Streamlining this process will not be easy because legislative changes and procedural changes would have to be made. Such changes are not necessarily desirable either because of built-in mechanisms to allow for participation by different groups in society.

We now outline the kinds of information necessary to respond to a specific criterion by a state 'EPA.' However, some of these functions might be the responsibility of other bodies in state government during the standard setting process.

EVALUATION PROCESS FOR A PROPOSED STANDARD

Preliminary Assessment. - The first step is a preliminary assessment of the problem. Once there is a federal initiative for a criterion for a substance, a state's first response is to ask: Could the substance be present? Generally available knowledge of types of industry, public treatment plants, solid waste disposal sites, sediment analyses and runoff obtained from various sources of information should be reviewed. This enquiry could conclude by a direct approach to industry for its information on the presence of the substance. Depending on the likelihood of the presence of the substance, a more-or-less extensive chemical monitoring survey could be undertaken.

Chemical and Analytical Methods. - If it is apparent that the substance is present at a level thought to be a problem, it is then necessary to gather detailed information on the chemistry of the substance in water. Up-to-date knowledge of the chemistry of the substance - dissociation, complexation, adsorption, chelation, solubility, volatility, as well as the effects of pH and redox potential on these phenomena has to be gathered and critically reviewed. Biologic information such as microbial metabolism and environmental fate, must be part of the review.

Information on the analytical methods for a substance must be gathered and studied. What forms of the substance are measured by a given method? Are the methods valid for the types of samples to which they are applied? Do they have sufficient precision and accuracy at the level of the standard? What are the economic problems of equipment, reagents, training of personnel and laboratory certification?

Toxicology. - The third step is gathering information on the toxicology of the substance. Ideally, what one really wants to know is whether an ecosystem can handle a certain input of the substance without changing significantly. But it is usually not possible to measure changes in an ecosystem accurately nor, to decide what is an acceptable change.

The actual data available from aquatic toxicology represents only an abstract model of what a pollutant could do in the environment. We test only a few representative, often 'domesticated', organisms, in ideal water (synthetic or especially clean), under physical conditions and with time factors not necessarily representing natural conditions. The end point selected is usually acute lethality, although effects on life cycle, reproduction, or other sublethal physiological effects are used. Ecological end points such as the effects on competition or energy transfer are little used.

Applicability. - The fourth step is deciding the applicability of the chemical and toxicological data to the local waters. Were relevant species used? Was testing done in water resembling that of the specific state with respect to temperature, alkalinity, hardness and other factors? The toxicological approach could be supplemented by ecological surveys after spills or comparisons of biota in streams high and low in the substance.

Treatability. - The fifth step is gathering information on the treatability, immobilization or neutralization of the substance. A study should include process evaluation for an industrial pollutant as well as waste treatment processes available for industrial and domestic pollutants. Possible treatment and disposal methods for solid waste sites, sediments or various non-point sources also need evaluation.

Enforceability. - The final point is consideration of all the above knowledge to decide whether the criterion is enforceable as a standard. If the criterion is realistic for local waters, can the substance be measured at the required level with sufficient precision? Further, are there realistic methods for reducing the loads of the substance entering (or in) the environment? The hydrodynamics of the receiving waters, the kinetics of the reactions determining the environmental fate of the substance and the loads of the substance, need to be known to answer this question.

Example of Cyanide

As our chief illustration we refer to a motion several years ago to revise the cyanide standard in Illinois. The activities included general assessment and some original analysis by the IEPA, general assessment and the holding of hearings by the Illinois Pollution Control Board, and a very extensive economic impact under the direction of the Illinois Institute of Natural Resources. For various reasons, mostly related to changes in treatment, industry wanted to change the basis of the standard from "total" to "free" cyanide.

No preliminary assessment of the problem was necessary because there was already extensive knowledge of the distribution of cyanide in Illinois waters from industrial monitoring and nearly 30 years of water quality monitoring. There was extensive knowledge of cyanide chemistry and chemical analysis. The key points in the testimony were related to the definition of cyanide, the pH dependence of complexation reactions, and the possibility of photodecomposition of the complexes in the environment. Because of a statistical study of the USEPA round-robin quality assurance results by our group in the IEPA and a thorough study of all available methods, it was concluded that methods for free caynide were not sufficiently reliable for use as a support for a standard.

The toxicity of cyanide was not a major issue because it was well known, but an expert witness was relied on during testimony to clarify the relation of the environmental chemistry to toxicity. The "free fraction" was regarded as the most toxic component. Because of the relatively high alkalinity and metal concentrations in Illinois waters, it was concluded that this fraction was probably quite small so that a standard of 25 ug/l was adequate to protect life for which other studies had not shown any adverse effects below 10 ug/l.

Treatability was an issue in that dischargers of strongly-complexed waste had more difficulty in meeting effluent standards than dischargers of "weakly-complexed" wastes. Since "strongly-complexed" cyanides were not believed to be treatable with conventional technology to the existing effluent limits, the dischargers were allowed a relaxation from 25 to 100 ug/l. They were warned, however, that they would have to convert to more exotic, more stringent methods of treatment if water quality standards of 25 ug/l were exceeded.

EVALUATION PROCESS FOR PRIORITY POLLUTANTS

For many of the priority pollutants, the procedure of standard setting will be more complex since there is little data on potential or actual occurrence of many of the organic compounds, we have initiated a chemical monitoring survey (alluded to above) and are developing a bioassay for accumulation and mutagencity determination. An industrial survey to learn of the presence of priority pollutants in processes has been conducted by the industrial permit section in connection with pretreatment plans.

There is extensive knowledge of the chemistry and analytical methods for metals on the priority list as these have been under regulation for sometime. Standards for these should perhaps be based on the form of metal actually present; however, analytical methods are still not adequate to do this.

In contrast, while there is little detailed knowledge of many organics on the priority list, analyses for these are not confused by the need to sort useful methods from a large number of methods which are available. The exact opposite is true: the analysis of many of the organic compounds is based on largely unvalidated protocols which use gas-chromatography/mass spectrometry as the detector. We have examined the theoretical problems associated with ascertaining the chemical composition of a complex environmental source. There are numerous problems associated with collecting an initial valid sample, for example (Tigwell and Schaeffer 1980a, b, c; Janardan et al. 1980a; Tigwell et al., 1980a, b), which are still overlooked in the Federal strategy.

We have addressed the question of whether or not the analytical data adequately reflects the actual composition of the source (Janardan and Schaeffer 1979a, Schaeffer and Janardan 1978, Schaeffer et al. 1980b) we found that these data are neither qualitatively (Janardan et al. 1980b) nor quantitatively (Janardan and Schaeffer 1979a) accurate desciptions of the sources from which they were obtained.

The toxicology of many of the priority pollutants is poorly known. There are inconsistencies in the original data sets as well as in the several standardization and extrapolation steps used by USEPA. The Science Advisory Board of USEPA has made an analysis of the propagation of errors of the process. This gives us some idea of the uncertainty in the number used as the criterion. We also have difficulties with the process. One is the use of sensitivity factors when the data set is large and representative. Also, the use of the sensitivity factor when the only data available is for a sensitive species, is unrealistic. Another concern is the irrelevance of the action level or tissue level number used in some cases for the residue limited toxicant concentration number.

Because of the lack of knowledge of the chemistry and the toxicology of these compounds, it is difficult to decide the applicability of the data to local waters. We have taken an initiative in this direction by trying to approximate local water conditions in toxicity tests with some of the conventional toxics. The results for copper suggest that a considerably higher level than allowed by USEPA's criterion is safe to native fish (Richey and Roseboom 1978). But it would be even more helpful if toxicity studies could be performed in 'natural' water samples (Schaeffer and Janardan 1975). This would probably require obtaining an LC_{50} in a number of water samples to establish a range of values rather than one 'true' value. We have also initiated an ambitious study comparing fish diversity and condition from Department of Conservation fish surveys with water quality information. We hope to be able to define better the conditions fish actually tolerate within natural waters.

We believe that treatment of priority pollutants will be especially complex. We have some preliminary data from some Illinois POTW's in which it appears that concentrations of some priority pollutants are changed little or none or even increased during conventional treatment (Kalman pers. comm.).

Thus, because the chemistry, toxicology and applicability of priority pollutants are still poorly founded, it will be difficult to decide on appropriate criteria for them. Problems with analysis and treatability suggest that such criteria as we have will be difficult to enforce.

We see several specific problems of enforceability in Illinois. Thus, there are still widely detectable levels of chlorinated hydrocarbon pesticides in the environment after a 7 year ban on their production or use. There are no realistic methods for treatment or neutralization of these substances. Another problem results from the fluctuation in feed stocks used in coking plants, since these feeds markedly alter effluent characteristics. Similarly oil refineries (and many industries) have intermittent effluents either because of batch process or fluctuating waste treatment. This presents serious monitoring problems.

CONCLUDING REMARKS

Our approach to these problems in Illinois has been substantially different than the federal approach. Building on the scientific base we have developed over the years, we initiated in 1977 a toxic substances program which is designed to address a large number of very specific questions which were formulated in advance. The data required to address these problems was characterized, and programs were developed which will provide the type, quantity, and quality of data needed to establish a data base which can be used for the purposes for which it was designed: ascertaining whether there could be an environmentally related cancer incidence in Illinois; ascertaining the possible substances involved; examining the distribution of such substances in the environment; developing procedures for using these data to estimate the probable impact such substances are having; ensuring that we have a data base adequate to support any regulations which are ultimately developed to control such substances, if regulations are needed.

We close with a brief discussion of toxicity and threshold, since the latter is at the heart of the human health effects issue, and because the existence of a dose response effect and a threshold will have to be ascertained from animal data, including aquatic life studies.

Janardan and Schaeffer (1977) and Janardan et al. (1979) presented a model (called the Lagrangian Poisson Distribution (LPD)) of the process leading to the production of chromosome damage by radiation and chemicals. One parameter of this model, g_2, is the ratio of the rates of the damage to the restitution processes, and at equilibuium it is the equilibrium constant (Schaeffer, et al. 1980d). Since the equilibrium constant is a thermodynamic parameter, the exposure-response of g_2 provides a thermodynamically based measure of the exposure response and, hence, of the threshold. The general relationship is given by Schaeffer et al. (1980e) in what they call the "filter model". This is written in linear form as:

$$F = b_1 \ln(D+1) + b_2(\ln(D+1))^2$$
$$F = R - Ro$$

where R is the response, Ro is the background rate at D=0, D is the exposure level, ln is the natural logarithm, and b_1 and b_2 are constants for a particular system determined from regression analysis. The first (b_1) term relates to the physiologic factors which reduce the exposure level as the substance is transported to the target organ. The second (b_2) term is related to the genetic factors which ultimately govern the actual amount of material (the dose) affecting the genetic material. Since the model is developed from a thermodynamically related response (g_2), thermodynamics tells us that only the beginning and end points of the process (i.e., physiology and genetics) are important in the partitioning of the exposure, and that there must be a (thermodynamic) threshold for the process. Without going into details which will be available in the literature, we present in Table 1 estimated thresholds for several of the priority pollutants and the estimated water quality criteria (which are developed from threshold estimates) needed to protect humans from adverse (induction of cancers) effects. It is seen that our estimates are substantially higher than USEPA's arbitrarily derived values.

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Table 1

Estimated water quality criteria for selected substances based on thresholds for cancer obtained by using the linear and whole form of the 'filter' model. Calculations based on USEPA methods.

Compound	Exposure Route	Threshold	linear only	ug/l whole	USEPA human health effects
*DDT	fish consumption	28 ppm	9.6x10-5	0.02	$1x(10^{-3}-10^{-5})$**
*Dieldrin	fish consumption	1.0 ppm	0.0046	0.1	$4.4x(10^{-5}-10^{-7})$
Dimethylnitrosamine	food and water	3.6ppm	369	1000	0.026-0.00026
+PAH's	food and water	0.29 mg/kg	7.72	10	$9.7x(10^{-3}-10^{-5})$

*bioaccumulates to a significant degree

**human health effects due to a risk of 10-5 to 10^{-7}

+To west of a number analyzed

ADEQUATE PROTECTION OF FISH AND WILDLIFE RESOURCES - THE FIRST DEFENSE AGAINST THE DETERIORATION OF HUMAN HEALTH AND QUALITY OF LIFE

Roger B. Griffith

U.S. Fish and Wildlife
18th & E Streets, N. W.
Washington, D. C. 20242

ABSTRACT

Water Quality Criteria and Standards result in regulatory decisions that have a direct impact on fish and wildlife resources. The continued loss of fish and wildlife resources resulting from pollution caused by man has precipitated the active involvement of the Service in developing and evaluating data for use in establishing water quality criteria and other pollution control standards. Within the past eight years, amendments to the Clean Water Act and other Federal laws have strengthened the Service's mandate and have stimulated it to become actively involved in the development of criteria and standards, and in providing comments on proposed state water quality standards.

While traditional water quality parameters such as dissolved oxygen, pH, temperature, and sediment continue to be of concern to the Service in the establishment of water quality criteria and standards, the primary concern is the introduction of xenobiotics into the environment, particularly the aquatic environment. The impacts of DDT on raptors, the impacts of PCB's in the Hudson River and the Great Lakes, and the loss of the Lake St. Claire fishery due to mercury contamination provide examples of the kind of occurrences that are the basis for the Service's concern.

Traditionally water quality criteria and standards have heavily emphasized human health and quality of life. This emphasis continues to be significant in current criteria and standards. Based on an evaluation of resource losses, the Service's position is that criteria, standards, and associated monitoring programs that are primarily designed to protect human health do not usually provide adequate protection to fish and wildlife resources. The exceptions are usually found where fish and wildlife are located in areas recognized as having a direct impact on human health. Recent events suggest that even human health is not adequately protected by the traditional approach. A more serious consideration of fish and wildlife resources would greatly reduce the loss of these resources and provide increased protection to human health. Through the review of water quality criteria and standards, the Service has developed recommendations to minimize the adverse impacts on fish and wildlife resources and provide an added margin of safety for humans.

INTRODUCTION

The U. S. Fish and Wildlife Service has a poster of a little boy with a crow perched on his shoulder pulling his ear. The poster caption is, "When wildlife tries to tell us something, we'd better pay attention." One may ask the question, "Does this illustration have anything to do with water quality standards"? The answer to this question is an emphatic yes!

Fish and wildlife are highly sensitive to environmental change. Decreasing water quality; altering stream temperature or flow; draining wetlands; channeling natural watercourses; clearing forests; and other changes that limit their food, water, or shelter; affect fish and wildlife immediately.

The composition and condition of fish and wildlife populations serve as a barometer of the quality of the environment for man. He too, must have clean water, fertile farmland, and healthy forests. Thus, proper use of our nation's land and water resources is in the interest of both man and the fish and wildlife resources he enjoys.

When important fish and wildlife resources are rendered unusable to humans because of pollution contamination, it is obvious that something is wrong with the environment. The gradual disappearance of desirable wildlife from a given area, in spite of legal and other overt protective measures, also indicates that something is wrong with the environment. Experience with such occurrences suggests that the effects on human beings may extend far beyond the loss of esthetic and recreational values.

DISCUSSION

The parameters measured by traditional and current water quality criteria and standards are easily quantifiable and generally physical or chemical in nature and provide the data for evaluating only select aspects of any aquatic system. Traditional criteria and standards do not address the impacts on the aquatic ecosystems involved nor the interaction between adjacent aquatic ecosystems, terrestrial ecosystems, and ecotones (zones of transition). They appear to be based on the assumption that man is external to the system that he impacts. However, the reality is that man is a part of the system he impacts and is vulnerable to the impacts he creates. This idea is part of the concept of an integrative framework linking human activity with the total ecosystem. This concept has been referred to as the ecosystem approach (IJC, 1978).

We believe that adequate protection of fish and wildlife and man requires political recognition of the ecosystem approach in developing pollution control mechanisms such as water quality criteria and standards. It is necessary to recognize that protected waters interface with the atmosphere, terrestrial ecosystems, other aquatic ecosystems, and ecotones to form a unified system. We must acknowledge the role of all living organisms including man: Clear recognition of the exchange of materials, such as atmospheric pollutants into and out of the system in biospheric perspective, is required. The ecosystem approach provides the philosophic basis for a view of man as part of nature and directs governmental efforts toward treatment of the patient (the Ecosystem) rather than the symptoms or disease. (IJC, 1978).

Long lasting xenobiotic pollutants and their impacts are a major reason for our interest in the ecosystem approach. Acid rain and the impacts of contaminants related to highway maintenance are additional examples of pollution phenomena better addressed by the ecosystem approach. Acid rain is an example of the interrelationship between air pollution, water pollution, and adverse land impacts which is not adequately addressed by the existing laws and regulations. The solution of airborne pollutants into rainfall adversely affects plant life and property and acidifies poorly buffered aquatic systems (IJC, 1979). Contaminants from highways provide classic examples of both point and nonpoint source pollution problems adversely impacting aquatic environments. One illustration would be the storage and use of road salts with cadmium based rust inhibitors that are washed into the adjacent streams.

Concern over these types of impacts has been expressed for several decades. A 1962 Michigan State University Extension Bulletin by Roelofs and Shick noted DDT impacts on nontarget organisms. Rachel Carson's Silent Spring was published the same year. In the sixties we and others proved that eggshell thinning in bald eagle, osprey, other raptors and fish-eating birds resulted from DDT and dieldrin exposure. (Ratcliffe, 1967; Hickey and Anderson, 1968; Porter and Weimeyer, 1969). We indicated that this phenomenon may be a major cause for the decline of these species. Their widespread dissemination in the aquatic environments was also confirmed.

More recently nonpesticide xenobiotics such as PCB's have had a significant adverse impact on human health, quality of life, and the environment. Of course early warnings were signaled by fish and wildlife impacts. The Minimata disaster in Japan was signalled by impacts on domestic cats. Apropos, the PCB laden carcasses of Coho salmon from Lake Michigan fed to ranch mink caused reproduction failure with similar health implications for humans.

Finally, acid rain, has caused its first obvious impact in this country through the decline of the fisheries in the high mountain lakes and streams in the Adirondaks and Appalachians. The European experience suggests that these effects may not be easily remedied.

These examples are of situations where the impacts could have been, or may still be, minimized by use of an ecosystem approach. This belief is incorporated into the Service's present position on water quality in general. On March 14, 1980, the Director of the Fish and Wildlife Service presented a statement before the House Committee on Merchant Marine and Fisheries on the issues of ocean disposal of polluted dredge material. He said, "...Our mandate directs us to conserve, protect and enhance fish and wildlife resources, but I would like to point out that we often learn about the damage some substance could have on human health from its effect on fish or wildife. These animals are not only valuable in their own right, they serve us well as indicators of the state of our environment."

This idea is not new, nor unique to the Service. Much of the background material used in developing this paper is based on position papers developed by such varied organizations as the International Joint Commission (1978, 1979, 1980), and the Wildlife Management Institute.

We became increasingly involved in the development of water quality criteria and standards and other regulatory actions because of the decline of fish and wildlife resources. Because of this involvement we have concluded that traditional regulatory approaches usually do not provide adequate protection for fish and wildlife since the classical parameters of water quality criteria and standards do not lend themselves to ecosystem impacts and as the major emphasis of such criteria and standards is on human health. The result is a system that only measures those quantifiable parameters that directly impact human health and quality of life and is rigorously applied only where human health is immediately impacted. The monitoring program associated with such standards is usually geared to react only when human health is endangered, and only likely to protect wildlife resources in areas considered imminently important to the protection of humans. As a result, the quality, and in some cases, the quantity, of wildlife resources have continued to deteriorate.

For example, several millions of dollars have been invested in developing commercial and sport fisheries in the Great Lakes. The concentration of PCB's in the fish flesh has limited the value of the Great Lakes Fishery as a commercial and sport fishery and impaired the reproduction of some species. Many important species are not considered safe for either human or pet consumption nor for use as feed for mink. The well publicized PCB problem on the Hudson River has impacted the fishery on that river as well as other associated wildlife. It's had far reaching impacts affecting humans that extend to New York Harbor and the New York Bight. Two bills have recently been introduced to address parts of the problem. One, introduced by Senator Moynihan, proposes to amend Title I of the Clean Water Act to redress the source of this problem at the expense of the Federal government. In Lake St. Claire, the fishery was closed because of mercury contamination. This occurred as a result of chloroaklyli waste discharges into tributaries of Lake St. Claire. Finally, Hooker Chemical was permitted to discharge approximately 900 lbs/day of unspecified chlorinated hydrocarbons into the Niagra River, a tributary to Lake Ontario. Lake Ontario commercial and sport fisheries were closed because of contamination by Mirex, a major component of the discharge. The impacts on fish and wildlife resources and quality of human life noted in these examples raises serious doubts regarding traditional approaches.

Our responsibilities under the Fish and Wildlife laws and treaties, and amendments to the Clean Water Act over the past eight years have fostered a closer relationship with our sister agency, the Environmental Protection Agency, regarding water quality criteria development and State Water Quality Standards review. Our objectives are to protect and enhance the propagation and production of healthy fish and wildlife populations. Hence, any action that would upgrade or downgrade present water quality standards or change classification and use are of interest to us.

Our preference would be to have standards and use classification developed in the context of an ecosystem approach. Since this may not be feasible at present because of cost concerns, we fully support the goals of the Clean Water Act, particularly the achievement, "of water quality which provides for the protection of fish, shellfish, and wildlife and provides for recreation in and on the water...by July 1, 1983." (P.L. 92-500). We believe that such an achievement is possible and practicable. Furthermore, the public costs of striving for this are less than the costs of reneging on such an effort. Case in point, the James River Kepone incident closed the river's commercial fishery in Virginia. Some predict the closure to last at least 20 years and estimate revenue loss to be in excess of $200,000,000 (1975 value). Human health costs resulting from this incident have not been fully calculated.

Thus, we advocate the constant upgrading of stream use classification systems to achieve the natural potential of the streams involved. We oppose the downgrading of use classification without regard for the stream potential for the purpose of relaxing water quality standards. We advocate the upgrading of present water quality standards to meet the use classification requirements and natural potential of the stream; and oppose the downgrading of standards where they cannot be justified by the existing use classification or natural potential of the stream.

Persistent xenobiotics in the aquatic system are a concern requiring adequate safeguards and monitoring. We advocate that no mixing zone be allowed for xenobiotics with an established or suspected latent toxicity such as polyaromatic hydrocarbons (PAH). Where compounds are stable, toxic, and known to biomagnify, no detectable limits should be permitted in the discharge. Measurable standards and effluent limits for compounds that biomagnify should only be established if a biologically measurable no effects level is known and a safety factor of at least 0.1 is used. An adequate biomonitoring system should also be required. Minimum mixing zone requirements for discharges of persistent material should not exceed 0.01 of the 96-hour bioassay LC/50. All water quality standards programs should make a serious effort to address these issues.

As a result of the above and other longstanding concerns, a technical meeting was held with EPA in the spring of 1978 to discuss these issues. The general categories used, and an updated summary of our position and rationale is as follows:

1) Adequacy of criteria with respect to implementation

The scientific credibility of criteria is of no value if the criteria cannot be implemented. Greater care must be utilized in blending these two requirements. The suspended solids and turbidity criteria in the current Quality Criteria for Water using the reduction of the depth of the compensation point is a scientifically sound criteria approach for turbidity but it cannot be effectively implemented as a viable enforcement regulation in many situations. In the case of suspended solids, the most significant impacts are abrasion and suffocation of the benthic community. Decrease of the compensation point is less important with respect to suspended solids.

2) Interactions of pH, dissolved oxygen (D.O.) and temperature

The special relationship of pH, D.O. and temperature in the aquatic environment needs to be considered. This interrelationship is synergistic in some cases, and can be synergistic to the actions of toxic substances, i.e. heavy metals and pesticides found in the aquatic environment. Criteria addressing the rate of change of these basic parameters and their interactions with other parameters are necessary (Sniesko, 1974; Davis, 1975). The pH can facilitate the toxic effect of metals. It can directly affect toxicity and rate of uptake for many organic substances. These relationships should be clearly defined in the criteria document. We believe that water quality standards for dissolved oxygen, pH, and temperature should be based on instantaneous readings rather than averages and apply to the entire water column if the aquatic biota are to be adequately protected. Furthermore, large rapid fluctuations of these parameters within the normal range of exposure can cause death due to the organism's inability to adjust. Such fluctuations need to be considered in developing standards.

3) The relationships of water column, streambed/water column interface, sediment, and interstitial gravel of streambed and water column.

The interrelationship of turbidity, sedimentation, and suspended solids in biological processes is not adequately addressed in the criteria. Sediment transport of toxic organics and metals is known to occur, as is accumulation by the biological community that bypasses the water column. The colloidal materials are also known to play a major role in the transport and reactions of toxic substances in the aquatic environment. Sediment concentrations of metals and toxic substances must also be monitored. EPA's approach has relied entirely on chemical criteria for the water column. Such an approach may not prove adequate for protection of fish and wildlife habitat which includes the gravel and sediment at the streambed/water column interface.

4) Endangered species considerations

Where endangered species are present, special considerations are required to preserve their habitat and prevent direct adverse effects from water quality changes. These considerations and requirements should be addressed in future criteria. Use of the Outstanding Natural Resource Waters Classification in water quality standards to protect endangered species and their critical habitat is one possible approach.

5) Biological monitoring

Consideration of the interrelationship between biological effects and the mechanisms of environmental distribution is necessary for protection of wildlife and requires a strong monitoring program which provides a continuous inflow of sound information. The documented tendency of the toxic organic and heavy metals to uptake from sediments directly into the aquatic biosphere is critical with regard to impact on fish and wildlife resources. Bioaccumulation monitoring is required to insure that the standards are meeting their intended goals with respect to fish and wildlife resources. In addition, monitoring should detect the impacts of altered use patterns and total discharge rates of highly toxic substances by industry, and others, that may modify the loading of the biological system in a given water body so as to require revision of criteria and standards. Guidelines for biological methods that can be used to derive effluent limits which can insure the biological integrity of the water including instream pathology, disease, stress, and behavior, need to be included in criteria development.

6) Chlorine, its byproducts and possible alternatives

Chlorine's immediate toxicity at low concentrations and potential adverse in situ interactions with organic material are cause for alarm (Jolley, 1975). It would appear that EPA's criterion for chlorine may not be stringent enough. The need for dechlorination and alternatives to chlorine disinfection is increasingly apparent. Seasonal disinfection, however, is not the answer since reports indicate that pathogens can survive for long periods in water (Coldwell, 1977).

7) Toxic substances evaluation and criteria.

We believe the treatment of toxicants in the criteria must be founded on an ecosystem approach. Application factors must be sufficiently stringent to insure ample protection against the impacts of bioaccumulation and chronic toxicity. Testing in the laboratory and field requires a community approach for adequate protection of fish and wildlife resources. Testing procedures also need to address behavioral impacts, synergistic interactions, food chain interactions, and the most sensitive species impacts relative to the total community. In setting standards, tissue residues should be an important consideration and safety factors should be introduced to compensate for uncertainty. The greater the uncertainty, the greater should be the required safety factor. However, maintenance of restrictive standards is expensive and will only minimally influence the aquatic environment. This situation underscores the need for more information on the relation of organism viability to water quality.

We have been involved with some activities where attempts have been made to address these issues and we have provided extensive comments regarding the approach to these issues in specific situations. In evaluating the Kepone pollution incident in the James River and Chesapeake Bay for example, we provided a proposed monitoring program addressing the biological uptake, persistence, redistribution, and fate of Kepone, and its metabolites in the aquatic ecosystem of the Lower James River. The proposal addressed the need for long term sampling of fish representing three trophic levels, benthos sampling, sediment, and some physical factors. A similar proposal was made regarding the ocean dumping of PCB laden dredged material in the New York Bight; a situation where we should have and will have input until a satisfactory solution is found. In addition, we questioned the impact of increasing the PCB laden discharge of the Fort Howard Paper Company into the Arkansas River in Oklahoma relative to meeting EPA's water quality criteria guidance and potential impacts on commercial and sport fisheries. Monitoring studies conducted by us indicated an increase in PCB levels downstream of the existing discharge. An evaluation of this problem by EPA and others is ongoing and may be part of the basis for a decision on amending the NPDES permit.

The foregoing indicates situations where our concerns raised valid issues which required serious appraisal in the administrative and regulatory processes regarding toxic substance evaluation.

Thus, the treatment of toxic substances in the current criteria and in most State standards makes us apprehensive. We believe that the maximum recommended criteria for any toxic substance should be no higher than 0.01 of the estimated 96-hour median tolerance for all indigenous species based on 96-hour LC50 data from bioassay tests, field studies, etc. In some cases, such as with PCB, 0.001 of the median tolerance limit is not sufficient protection.

8) Compartmental or Residue dynamics.

Compartmental dynamics is used in toxicology to define how an environmental concentration of a toxicant, its residues or its metabolites, develops complicated interactions affecting the physiology and biochemistry of various organ systems in an animal. In an ecosystem, similar interactions occur between the various living and physical components of the system that may be defined by the same terms. We must be aware of the ultimate fate of a contaminant. We must know the toxicological significance of the period of exposure to the contaminant, its residues, and breakdown products in a trophic level. Such an approach would be consistent with the requirements used in regulating the deliberate contamination of animals and other food products by pesticides or drugs. To date this concept appears to be absent from the process of developing water criteria. Such a concept is essential to a total ecosystem approach and is necessary to understand how food chains magnify and move substances, particularly at the freshwater, estuarine, and ocean interfaces. It could have considerable impact on predicting the potential adverse environmental effects of chemical pollutants.

9) Mixing zones

The regulatory concept of a mixing zone is adverse to the well-being of fish and wildlife resources. Although we recognize the necessity of permitting mixing zones under some conditions, nevertheless, these zones must be defined in a manner that fully accounts for the above concerns such as excluding mixing zones for toxic substances. This is especially critical relative to adverse effects on migratory species of fish, spawning, embryo development, and aquatic habitat. Guidelines in the criteria need to be developed accordingly.

A mixing zone definition must be based upon the behavior of chemicals in aquatic environments and consider such things as biological uptake and endangered species habitat contamination. In the review of criteria and standards, we have developed minimum mixing zone specifications to meet these concerns. The specifications are:

> Location: Biologically important areas are to be identified and protected. In particular, mixing zones should preserve a zone of passage for migratory fish and other organisms in a water body.
>
> Size: The area and volume of an individual zone or group of zones must be limited to the areal size and volume that will not interfere with the designated uses or with the established community of aquatic life in the area for which the uses are designated. Growth projections should be taken into account in setting the total mixing zone for the water body.
>
> Shape: The shape of a mixing zone should be a configuration that is easy to locate in the body of water and that avoids impingement

on biologically important areas. A circle with a specified radius is generally preferable, but other shapes may be specified in the case of unusual site requirements. "Shore-hugging" plumes should be avoided.

Outfall Design: Prior to designating any mixing zone, the State should assure that the design and location of the existing or proposed outfall provides maximum protection of aquatic resources.

In-zone Quality: Water quality standards should provide that all mixing zones conform with the following requirements. Any mixing zone should be free of point or non-point sources related to:

(a) materials in concentrations that exceed 0.1 to 0.01 96-hour LC50 for sensitive biota significant to indigenous aquatic community;

(b) materials in concentrations that settle to form objectionable deposits;

(c) floating debris, oil, scum and other matter in that form nuisances;

(d) substances in concentrations that produce objectionable color, odor, taste or turbidity; and,

(e) substances in concentrations which produce undesireable aquatic life or result in a dominance in nuisance species.

Little progress has been made in addressing the concerns that we expressed at the 1978 meeting. We have not seen an overt active effort to rectify the issues we have raised in a manner that will provide adequate protection of fish and wildlife resources. We hope to see more active efforts along these lines in the future for the protection of human quality of life as well as fish and wildlife resources.

SUMMARY

In summary, the Service advocates the ecosystem approach to regulation including water quality criteria and standards. We are concerned with the criteria and standards because they result in regulatory decisions which directly impact fish and wildlife resources. Pollution related loss of fish and wildlife resources continues while water quality criteria and standards have, for the most part, emphasized human health and quality of life. A more serious consideration of fish and wildlife resources will provide increased protection of humans. To alleviate adverse impacts on fish and wildlife resources, we advocate the upgrading of standards and use classifications to address the natural potential of the stream. We recommend that the introduction of xenobiotics into the environment be accompanied by adequate safeguards and monitoring. Further, we advocate no-mixing zone for persistent stable xenobiotics with latent toxicity, such as PAH, and have outlined some mixing zone specifications. Recommendations are also proposed in other general categories: interaction of pH, D. O. and temperature; suspended solids, sediments, and the water column/streambed interface; chlorine; endangered species; and implementation of criteria and standards.

We advocate that criteria be developed which include consideration of:

1) Data from bioaccumulation and biomonitoring studies.

2) The impacts of pollutants concentrated in the sediment.

3) The additional loading resulting from pollutants entering the water column, or aquatic ecosystem, through the air-water interface and streambed/ water column interface.

4) The special interrelationship of pH, dissolved oxygen, and temperature to the aquatic environment and aquatic life.

5) Approaches and alternatives to chlorination which minimize its adverse effects on the environment.

6) The long term impacts of xenobiotics in the aquatic

ecosystem including their compartmental or residue dynamics.

7) Enforcability of standards based on the criteria.

Further, we advocate that standards be developed that have:

1) Sound implementation mechanisms.

2) A clear detailed mixing zone policy that adequately considers xenobiotics.

3) Dechlorination as a part of the chlorine use requirements.

4) Requirements to upgrade use classification in keeping with the potential of the water body.

5) Protection of existing clearn water and unique or ecologically critical areas, ie. endangered species and their habitat by Outstanding Natural Resource Water Classification.

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MONITORING, SAMPLING AND ANALYSIS TO SUPPORT CRITERIA AND STANDARDS

SECTION V

A ROLE FOR STATISTICS IN WATER QUALITY STANDARDS?

Robert C. Ward, Jim C. Loftis, Gary M. Smillie
and Thomas G. Sanders[1]

Agriculture & Chemical Engineering Dept.
Colorado State University
Ft. Collins, CO 80523

ABSTRACT

Water quality monitoring, a statistical sampling program, is used by water quality management agencies to evaluate compliance with standards--both stream and effluent. Stream standards are often expressed as water quality extremes not to be exceeded if a designated use of the stream is to be protected. The rather absolute nature of stream standards precludes use of the probabilistic monitoring results to check compliance in a statistically sound manner.

A review of monitoring and stream standards relationships within a regulatory water quality management agency are presented. The advantages of expressing standards in terms better related to monitoring results are noted. The evolving nature of water quality management and the embryonic state of development of water quality hydrology, the sciences needed to properly support management efforts, are discussed.

Now that much of the "rush" to establish the large water quality management efforts (e.g., construction grants program, 208 planning program and NPDES) is over, it is hoped that the "in-place" management system will be permitted to mature and develop the capability to better utilize water quality data and information in its decision-making, particularly, with respect to evaluations of stream standard compliance and ambient water quality trends.

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[1] Respectively, Professor and Assistant Professor, Department of Agricultural and Chemical Engineering, and Graduate Research Assistant and Associate Professor, Department of Civil Engineering, Colorado State University, Fort Collins, CO 80523.

INTRODUCTION

Water quality standards are the basis for water quality management in the United States. In order for standards to be effective, information which is only obtainable through a monitoring program is needed to determine compliance of water bodies with their standards. Standards, as they are assigned to a specific reach of a stream, are generally fixed quality limits which define a water's acceptability for a given use. On the other hand, water quality monitoring or sampling must be viewed in a statistical context. Consequently, the statistical nature of monitoring data often relate poorly to the fixed limits of standards.

The purpose of this paper is to examine the role statistics could play in reducing or eliminating the disparity between fixed limit standards and the probabilistic nature of water quality data. The examination is performed by reviewing: 1) the advantages of having statistics in current standards; 2) the probabilistic aspects of water quality; 3) the administrative problems faced when uncertainty is incorporated into standards; and 4) potential ways to reconcile the disparity between standards and monitoring without creating administrative problems.

DISCUSSION

PROBLEMS ENCOUNTERED IN ESTABLISHING STANDARDS

Prior to the requirements by PL89-234 (the Federal Water Quality Act of 1965) that stream standards be used in all state water quality management efforts in the United States, there were a number of different approaches used to incorporate the concept of a standard into water quality management. Many of these pre-1965 approaches made use of the water quality criteria developed in California (State of California, 1952; McKee and Wolf, 1963). These criteria reports were very comprehensive reviews of the literature describing a range of effects of various water quality levels on various uses. The judgment as to the level of quality (i.e., criteria) was most applicable to a given situation (use and other local factors) was left to the person dealing with the situation and the management strategy chosen.

PL89-234 required the use of stream standards in all states and, thus, created a need for detailed information on which to base standards development. The law also specifically stated in Section 10c(3):

> "Standards of quality established pursuant to this subsection shall be such as to protect the public health or welfare, enhance the quality of water and serve the purposes of this Act. In establishing such standards the Secretary, the Hearing Board, or the appropriate State authority shall take into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other legitimate uses."

Thus, the "use" of the water was the basis for developing standards. To provide a common basis across to the United States for establishing standards, a new set of water quality criteria were developed (NTAC, 1968) where the best available information and scientific judgment was used to arrive at a set limit (criteria) which, if exceeded, would render the water unfit for its designated use.

These limits have not been established without considerable debate. This debate has resulted for several reasons which may be categorized as follows (Clark, 1967; and Krenkel, 1979):

1. Inadequate knowledge on environmental effects of many constituents.
2. Antagonistic and synergistic reactions among the constituents.
3. Analytical difficulties, and
4. Complications due to the stochastic nature of water quality.

Criteria can be formulated by assuming that there are no antagonistic or synergistic effects and using the current state of the art with the best available scientific judgment and safety factors. Standards can be developed using the criteria and enforced assuming there are no analytical difficulties or uncertainties and the water quality constituent concentration does not behave in a probabilistic or random manner.

The above is done in order to provide a framework for water quality management. The basic problem, that would affect any framework chosen, is the lack of knowledge. Clark (1967) and each of the criteria books published since PL89-234 (NTAC, 1968; NAS/NAE, 1973; and EPA, 1976) have noted limitations associated with establishing criteria while stating the need to continuously upgrade water quality standards on the basis of available information. Wolman (1969), stating this another way, noted "the development of criteria for the protection of health has invariably preceded full scientific understanding and acceptance."

PL89-234, and subsequent federal laws, have demanded that water quality management use standards based on quality levels needed for a given use of the water. With incomplete knowledge, the only approach available was that chosen--assume no antagonistic or synergistic effects, use the best available information and judgment, and deal with the analytical difficulties and hydrologic variations during implementation or enforcement of a standard, not in the standard itself.

Criteria and standards expressed as set limits or extremes in quality, in general, do relate well to defining a quality acceptable for a given use. This, as noted above, has been the general purpose of criteria and standards and those of today (e.g., EPA, 1976) serve this one purpose well. The "use of water" is a very convenient way to organize a large body of data and present it in an easily understood format.

Spot Checks

The monitoring established to check compliance of stream standards has primarily been based on "spot checking" (Sherwani and Moreau, 1975). A sample is collected, analyzed and the results checked against the limit defined for that reach of stream. The standard, at that instant in time, either is, or is not, being violated. The data, beyond this spot check, often has no further use to the agency relative to stream standard compliance. Given the times and areas not sampled, it is difficult to generalize the conclusions much beyond the immediate yes/no statement using the "spot checking" approach.

The NAS (1977) report on environmental monitoring noted this "spot checking" approach being used by the Council on Environmental Quality in lieu of a better approach. Quoting the NAS report,

> "...the Council on Environmental Quality (CEQ), in their Sixth Annual Report (CEQ, 1976), used what data were available from records in STORET (the EPA ambient water quality data bank) to determine the percentage of times ambient standards had been exceeded."

Means and Ends in Monitoring

The "discontinuity" between standards, the basis for water quality management, and water quality monitoring have created some problems in the monitoring programs. As with any large organization where goals, or the means of reaching the goals, are unclear, the means often become the ends.

Sewell (1975) notes the following:

> "Means become confused with ends because we have failed to define ends. The processing of reports becomes an end, not a means. Assumptions are forgotten and purposes are distorted to avoid disrupting the organization of technicians."

The poor relationship between standards and the results of monitoring and the poorly defined procedures for statistically analyzing water quality data imply a poorly defined "end." Consequently, the means, the monitoring itself, becomes the end.

To change this situation will involve defining a quantitative end. The reason this has not been done in the past stems from the extremely complex problems involved in relating ambient, fixed-station (rather low freqency) sampling to standards expressed as extremes in quality not to be violated and in using low frequency data from a stochastic process to define statistically conclusive trends.

If assumptions are made to render the problem tractable, the solutions are deemed questionable due to the assumptions. If the assumptions are not made, the problem simply cannot be solved with available data and current training of the majority of monitoring personnel, if they can be solved at all.

As the NAS (1977) report pointed out, there are simply not enough people in environmental management, particularly water quality management, who have the background to properly define the "ends" of monitoring and develop the operating procedures (means) to get there. Monitoring is a statistical operation whose design requires an understanding of both statistics and water quality hydrology as well as the organizational limitations (criteria, standards, permits, economics, personnel, operating schedules, etc.).

PROBABILISTIC CONSIDERATIONS

Water quality is a stochastic process and its monitoring must account for this fact through the use of statistical principles. Standards, as noted above, do not acknowledge this fact. The uncertainties associated with the stochastic nature of water quality are usually dealt with in the implementation of the standard, not in the standard itself.

Such an attempt to account for the stochastic nature of water quality, deals, perhaps, as much with a lack of knowledge on water quality as a random variable as it does with potential problems created by initially including uncertainty of any kind in a standard. While many engineering designs (highways, dams, drainage channels, etc.) are based on information derived from stochastic hydrology, there rarely is a need to "enforce" a law based on the findings of stochastic hydrology.

Thus, even though it is possible, and perhaps highly desirable, from a technical management viewpoint, to consider water quality as a stochastic process, especially in terms of gaining as much information as possible from a water quality monitoring program, the incorporation of an uncertainty, of any type, into a legal standard may render that standard unenforceable from a practical point of view.

Water Quality as a Stochastic Process

Reconciling the need for enforcement of a strict, certain regulation with the statistical (often uncertain) results of monitoring, has been alluded to a number of times in the literature. The call for consideration of water quality as a stochastic variable has appeared more frequently as the field of stochastic hydrology becomes more established and useful in day-to-day life in the United States.

Ewing (1970) notes the great progress being made in the field of stochastic hydrology and the increased precision being incorporated into predictions of water quantity relationships due to probabilistic considerations. He further states:

> "The progress made in recent years in considering the stochastic nature of water quantity relationships should provide a convenient point of departure for application of probabilistic considerations to water quality relationships. Environmental scientists have done far too little to take advantage of this lead provided by hydrologists."

Pertaining directly to water quality standards, Ewing (1970) states:

> "The establishment of state water quality standards for both interstate and intrastate streams has recently been accomplished. In practically every case, DO requirements have been set without any reference to the probability of these levels being exceeded. It would seem that the state-of-the-art is rapidly advancing to the point, however, where the probabilistic concept should be recognized more specifically in the statement of the water quality standards themselves."

Roberts (1976), noting the stochastic nature of water quality, states:

> "It is difficult or impossible to develop a pollution control program which can ensure that any reasonable ambient standard will never be violated."

Swainson (1976) in discussing the technical problems that water quality management institutions must deal with notes:

> "Furthermore, they must be able to respond to the uncertainties associated with some water quality conditions and to the stochastic character of water quality itself."

Montgomery and Hart (1974) note that water quality criteria should include statistical considerations when they state:

> "Quality criteria are usually expressed as concentrations but it can be argued that a more satisfactory approach would be to define a statistical probability of a given concentration occurring."

Drechsler and Nemetz (1978), more specifically, state:

> "In order to have an effective regulatory system, it is essential to know:
>
> a) the type of statistical distribution from which the sampling data is derived, and
>
> b) the location of this underlying statistical distribution in relations to the government standard."

Clark (1967), in noting problems in standards created by hydrologic variations, cautions:

> "A statistician undoubtedly would argue for setting standards only in terms of probability...This involves a risk factor--a concession that standards will not be met 100 percent of the time."

This last point, perhaps more than any other, complicates the incorporation of probability into the standards that must be legally enforced by a regulatory agency.

Risk and Uncertainty

If water quality is to be considered in a probabilistic manner, the concepts of risks and uncertainty, from a water quality perspective, need to be clarified. In regulatory water quality monitoring, there is always a risk that a given standard will be violated. This risk is a property of the random nature of water quality. It is a fixed property, it does not change as more data is collected. If the water quality distribution were known exactly, then the risk that a standard will be violated would be known exactly. In water quality monitoring risk is the probability that a specified standard will be exceeded within a specified time period. Risk is inherent in the phenomenon of water quality and cannot be avoided.

Uncertainty, on the other hand, stems from our inability to know exactly what are the properties of the underlying population. To determine, actually, to estimate, these properties we use data which are often deficient or biased and statistical techniques which are weak. This results in errors and losses of information which cause us to be uncertain of the true population properties. Uncertainty is, thus, the difference between a population property and an estimate computed from sample data. Uncertainty can be decreased only by collecting more or better data and using better statistical techniques.

Computing confidence limits on means or extremes in water quality helps evaluate the uncertainties in the data and its statistical parameters.

ADMINISTRATIVE NEED FOR FIXED LIMITS

With regulatory water quality management attempting to legally enforce standards, the concepts of risk and uncertainty associated with sampling a stochastic process can only complicate the enforcement actions. It is perhaps better, or easier, from an enforcement viewpoint, to not incorporate these concepts into legal aspects of management. McKee and Wolf (1963), in referring to this problem in a rather broad context note:

> "There is a tendency...for regulatory authorities to promulgate standards of questionable scientific justification to serve as a crutch that facilitates administrative action and enforcement."

Standards, stated in fixed limits, do remove much of the uncertainty from water quality management--the standard either is, or is not, being met! Even so, legal enforcement of the fixed limit standards is often challenged by attacking the judgments needed to set the limit in the first place.

Wolman (1969) summarized the problem as follows:

> "Administrators are constantly searching for mathematical certainty in the solution of many of these complex problems, despite the fact that there is a high degree of uncertainty in the underlying scientific principles that are involved in establishing such standards."

Montgomery and Hart (1974), although stating statistical considerations can be argued in criteria, also note that from an enforcement viewpoint, the incorporation of statistics would be difficult to implement. If this were done, the statistical nature of the standard would be the basis for argument. The judgments used to arrive at the standard would no longer be the major point of contention. Roberts (1976) states this point as follows:

> "Unfortunately, a probabilistic rule is almost assuredly not administratively feasible. Determining when such a rule is violated is a matter of statistical inference. One can just picture the experts and lawyers employed by various parties trying to agree on what tests to use in different circumstances!"

The probabilistic aspects, needed in standards to better match the monitoring programs, are often viewed by water quality managers as opening the door to more pollution while those being regulated, likewise, view the uncertainty in regulations as an opportunity to devote less resources to environmental control. An example of bending a more flexible system to better advantage is found in Berthouex (1974), in a paper discussing statistical effluent standards. Berthouex (1974) recognizes that variance in effluent standards consist of the sum of three variances:

$$\begin{aligned}\text{Variance (Reported Data)} &= \text{Variance (Process)}\\ &+ \text{Variance (Laboratory Analysis Tests)}\\ &+ \text{Variance (Sampling)}\end{aligned}$$

Recognizing this, he states:

> "Since the regulatory agency looks at the reported data, it is in the interest of the treatment agency to present their process in the best light by minimizing variance due to analytical procedure and due to sampling. This permits somewhat more variation in actual process performance and should lead to some economics in operation."

With more statistics in standards, how many other statistical twists could be found to further increase the "economics of operation"?

With situations like that above, it is easy to see why more strict legal standards are maintained even though the ambient monitoring being performed has little relationship to the standards. Strict, fixed-limit standards relate better to intensive, enforcement-oriented studies and monitoring which is intensive enough to relate well to the intensive data demands implied by enforcement of standards.

With current management strategies, where the legal standards (the only water quality goals currently used) do not relate well to ambient monitoring, the use of ambient monitoring data within a water quality management organization is restricted or limited. This fact has been recognized by EPA's (1977) Standing Work Group on Water Monitoring, and they recommend a shift in monitoring resources away from ambient monitoring to special survey monitoring where the objectives of the legal enforcement purposes of management are much better related to the monitoring performed. If the management strategy cannot be changed, this is a way to better use monitoring resources.

Ambient monitoring, in this case, is left with only a goal of detecting trends in water quality. However, if summaries of information on streams in compliance with their legal standards is wanted (and it is by law, 305(b) reports are an example) then the ambient data is again asked to provide information on standards violations. Thus, even though specific enforcement actions may not be based on the relationship of ambient monitoring to standards, the overall evaluation of the success of environmental management, as evidenced by improvement or decline in water quality, is often expressed numerically as miles of streams not in violation of legal standards.

COMPLEMENTARY MANAGEMENT TOOLS

The foregoing discussions have noted both sides of the argument relative to including statistics in water quality standards. Standards are generally used in water quality management for a number of different purposes--from legal enforcement, to planning management priorities, to writing effluent permit limitations. Perhaps standards, as they are currently written, are being asked to serve too many diverse management purposes. And if this is the case, they may be serving one role, legal enforcement, well while not serving other more technical, or planning oriented, roles to the level needed for an effective and efficient total management program.

Water quality management agencies are complex organizations dealing with law, planning, engineering, chemistry, hydrology, biology, statistics, etc. For all of these disciplines to mesh well into a smoothly operating system, there must be means for all disciplines to relate to the goals of the organization.

Perhaps, as water quality management continues to evolve, it is time to consider expressions of water quality goals (or requirements, or objectives or guidelines, etc.) that stem from the current legal standards, but which relate more directly to the behavior (biology, hydrology, etc.) of the system being managed. For individual streams, water quality guidelines could be developed which are based on standards yet account for the stochastic nature of water quality hydrology and additionally have meaning in terms of management's use of monitoring data.

Such guidelines (or goals, objectives, requirements, etc.) could be developed in a number of different ways. Sanders and Ward (1978) discuss the use of classical quality control strategies as a potential framework for assisting water quality managers in better utilization of monitoring data. Flemal et al. (1979) describe the types of information and analyses that can readily be obtained to assist in evaluating water quality standards and their revision.

Currently, the authors are examining additional ways of relating monitoring to the goals expressed by current water quality standards. Approaches being examined include: 1) determining the influence of the 96-hour time frame of a 96-hour LC_{50} criteria, for example, may have upon the sampling frequency of a monitoring program; 2) determining the percent of time standards are not in effect due to low flow and other exclusions; and 3) computing the frequency distributions for water quality variables and noting the statistical confidence in the cumulative distribution function available in the area of the existing standard. These activities are designed to better relate information obtainable from current monitoring efforts to current monitoring efforts to current standards. These standards may remain at their extreme value for enforcement purposes. However, in the future additional guidelines must be established which are more fully cognizant of the stochastic nature of water quality hydrologic processes and are thus better able to serve the informational needs of the management organization itself.

AN EVOLVING PROCESS

Water quality management in the United States has progressed substantially over the past 15 years and it continues to evolve today. It must evolve as the sciences (e.g., biology, hydrology, chemistry, etc.) that support its activities become more refined and precise.

As the knowledge of water quality as a stochastic process builds, this knowledge should be utilized in refining management strategies, either through stating the goals of standards as guidelines to be used as management tools or by direct incorporation of statistics into the standards themselves. By making better use of the developing science of water quality hydrology, water quality management, in general, and water quality monitoring, specifically, can be based upon a stronger footing. If some means within water quality management is not provided to utilize the science of water quality hydrology, management loses much of its effectiveness.

Ruff (1978) notes that there are problems involved when a regulatory agency attempts to constantly change its regulations in light of new information (e.g., new information from the science of water quality hydrology).

> "If...the regulations themselves are changing continually in pursuit of an evolving goal or in light of unfolding information, enforcement--in the sense of using the law to compel compliance with a set of well-defined and court-tested regulations--becomes impossible."

Thus, to be realistic regarding use of water quality hydrologic principles, the standards themselves may not be changed. The goals they represent will, however, be analyzed, interpreted and expressed in terms directly related to the water quality processes being managed and monitored.

SUMMARY AND CONCLUSIONS

Water quality standards are generally based on extreme quality values while regulatory water quality monitoring is generally more suited, from a statistical viewpoint, to measuring measuring means in quality. Nevertheless, the "spot checking" approach to monitoring has been chosen to check standards compliance. With the results of monitoring so poorly related to the goals of management, the means of monitoring often become the end.

As the concepts of stochastic hydrology become more generally accepted, the possibility of their use in water quality management increases. This would involve viewing water quality as a stochastic process and utilizing risk and uncertainty in water quality management. Regulatory water quality management, being enforcement oriented, cannot include uncertainty in its standards or regulations without further opening the door to numerous debates and lawsuits regarding strategies chosen to manage water quality.

The stochastic nature of water quality can, however, be incorporated into the implementation of standards, program planning, reporting of water quality conditions, and management decision-making. This would greatly enhance the scientific basis for management and make such management more precise.

Water quality management in the United States has progressed through some rather hectic times during the past 15 years as 208 planning, construction grants and NPDES programs had to be implemented within stringent deadlines and often without the information needed to scientifically justify the decisions made. Now that the "rush" to establish these programs is over, it is hoped that water quality management programs will be permitted to mature and develop the capability to better utilize the information currently developing in fields such as water quality hydrology. This is particularly true with respect to the use of monitoring data and information in water quality management decision-making (e.g., evaluations of stream standard compliance and ambient water quality trends).

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ENVIRONMENTAL CONCENTRATIONS
OF
BUTYL BENZYL PHTHALATE

Paul R. Michael, William J. Adams, William E. Gledhill,
Orville Hicks, Robert G. Kaley, II

Monsanto Company
800 N. Lindbergh Boulevard
St. Louis, Missouri 63166

ABSTRACT

Butyl benzyl phthalate (BBP)is a large volume plasticizer used in the production of flooring materials and other household products. It has been included as one of the 129 priority pollutants and is one of the chemicals for which water quality criteria and standards are being set. To determine its presence in environmental waters, we periodically conduct monitoring programs of rivers, lakes, and streams. Sampling is conducted to obtain water and sediments that are representative of the water body. For analysis, water samples are extracted with methylene chloride and the extracts are concentrated in a Kuderna-Danish evaporative concentrator. Sediments are extracted with cyclohexane. The extracts are analyzed for BBP by GC/MS in the selected ion monitoring mode. Recoveries from spiked water average 90% with a detection limit of 0.5 ppb. Average recoveries and detection limits in sediments are 116% and 100 ppb, respectively. Results to date indicate that aqueous concentrations average less than 1 ppb and that concentrations in the sediments are less than 100 ppb.

The significance of these results can be understood only when viewed in the light of available aquatic toxicity data. Since BBP is chronically toxic to Daphnia and fathead minnows in the 100-800 ppb range, a 100-1000 fold safety factor currently exists in environmental waters. Analytical monitoring data together with aquatic toxicity information should be the major factors in setting water quality criteria and standards for individual compounds.

Butyl benzyl phthalate, whose structure is shown in the first slide, is produced by Monsanto Company under the trade name Santicizer® 160 (slide 1). Production in the U.S. is in excess of 100 million pounds per year (slide 2). The ester's chief use is for plasticizing polyvinyl chloride plastics, although it is also used with a variety of other synthetic resins. It is commonly found in vinyl flooring material and a number of other household products. The uses do not involve discharge into the environment in large quantities similar to pesticides or detergents. However, butyl benzyl phthalate may enter the environment during manufacture, transportation, processing, or by leaching from finished materials. Because of the wide spread use of phthalate esters in general and the reported high aquatic toxicity of one ester in particular, EPA is currently developing water quality criteria and standards for six of the esters, including butyl benzyl phthalate. Today I will discuss the analytical techniques we have used to measure butyl benzyl phthalate in the environment. I will also discuss the results we have obtained from sampling a number of different environmental sites over a two year period.

In order to protect aquatic organisms, we have to be concerned that levels in the environment are below levels which cause toxic effects. The subject of this paper is the toxicity and exposure of butyl benzyl phthalate to aquatic organisms.

The results of a number of acute toxicity tests we have conducted are shown on the next slide (slide 3). It is apparent that this ester is acutely lethal to several common species in the 1 to 5 parts per million range. The two values listed for fathead minnows are for hard and soft water. We selected two representative species, Daphnia and fathead minnows, for chronic toxicity testing. The results on the next slide indicate similar values for both species (slide 4). Butyl benzyl phthalate becomes chronically toxic at the level of several hundred parts per billion.

Our analytical techniques were chosen with these levels in mind. We wanted to be certain that our sensitivity, precision, and accuracy were more than sufficient to measure the compound in the aquatic environment at concentrations well below the chronic toxicity levels. The scheme we used for analyzing water is a fairly typical trace organic analytical procedure (slide 5). Aqueous samples were usually one liter in size and were extracted three times with methylene chloride. The extracts were dried with anhydrous sodium sulfate and concentrated to approximately one milliliter in a Kuderna-Danish evaporative concentrator. The extracts were analyzed for butyl benzyl phthalate by GC/MS in the selected ion monitoring mode.

For sediments, ten gram samples were weighed into polyethylene centrifuge tubes along with 10 ml of cyclohexane and 10 ml of deionized water (slide 6). The extraction was performed for 30 seconds using an ultrasonic homogenizer. The mixture was centrifuged to separate the phases and to remove suspended particles. The cyclohexane was concentrated to one milliliter and analyzed by GC/MS.

We have used GC/MS as a means of detection for all of our work because of its combination of selectivity and sensitivity. The next slide contains an electron impact mass spectrum of butyl benzyl phthalate (slide 7). You can see that the two most intense ions are at m/z 149 and 91. M/z 149 is characteristic of most phthalate esters. However, an intense ion at 91 identifies this as a benzyl phthalate. For maximum sensitivity we use the selected ion monitoring mode in which we monitor the intensities of only a few ions during the GC run instead of continuously scanning the entire mass range.

The next slide is a selected ion monitoring chromatogram of a composite phthalate ester standard containing both dialkyl and alkyl aryl phthalates at 0.4 to 0.8 mg/L (slide 8). Butyl benzyl phthalate is easily distinguished using three characteristic features: an ion at mass 149, an ion at mass 91, and the correct GC retention time of 5.5 minutes. The next slide is an SIM chromatogram of an extract from a chemical plant effluent (slide 9). The extraction procedure was similar to that described earlier. The level of butyl benzyl phthalate in the plant effluent was measured to be 78 ppb and you can see that we are getting a pretty healthy signal for both ions. To show you that we are not pushing our limit of detection for this method, we have only concentrated the extract to 10 ml and injected one microliter. By concentrating the extract to 1 ml and injecting 5 µl, we can determine quantities at the sub-ppb level. Looking at the slide, it might appear that butyl benzyl phthalate is the only compound present. I can assure you that this is not the case. If any of you have ever seen a chromatogram of a typical plant effluent using a non-specific detector such as a flame ionization detector, the chromatogram is usually very complex. That is, it has a lot of peaks. The absence of other peaks in this chromatogram is due to the selectivity of the mass spectrometer as a GC detector.

To adequately define our precision and accuracy at these low levels, we have developed a rigorous quality assurance program for method validation outlined on the next slide (slide 10). First the instrument is calibrated to determine its linear dynamic range. Each standard is injected in triplicate. Once the operating range has been established, all samples must fall within this range. Those more concentrated must be diluted and those too dilute are either concentrated or listed as none detected. To determine the accuracy and precision of the entire method, we spike aliquots of representative river water and sediment with known amounts of butyl benzyl phthalate. Five replicates are prepared at each level, including a set of blanks for both river water and sediment. The results of our recovery experiments are shown on the next slide (slide 11). You can see that our recovery or accuracy is quite good at all levels and our precision is certainly acceptable considering the quantities for which we are looking. Note that the levels in water are in ppb and the levels in sediment are in ppm.

The detection limit in sediment is higher primarily because we are extracting only 10g of sediment vs one liter (1000g) of water. In general, our detection limit is limited by interfering species and laboratory blanks. As I mentioned before, butyl benzyl phthalate is a widely used plasticizer and it is very easy to pick up contamination during sample workup and analysis. However, the levels I have listed here are ones that we can accurately and reliably determine in our laboratory. These detection limits are more than adequate to establish the lack of toxic effects to aquatic organisms. The detection limit of 0.5 ppb in water is several hundred times lower than the chronic toxicity of butyl benzyl phthalate to Daphnia or fathead minnows shown in slide 4.

In any effort to determine environmental concentrations, the sampling phase of the program is just as important as the analysis (slide 12). Samples were collected to provide a good representation of a given body of water. Water samples were composites of the entire water column. They were taken by lowering a Teflon® tube to the bottom of the river or harbor and then slowly raising the tube to the surface while pulling water through it at a constant rate. Sediments were taken with Ponar and Student-Ekman grab samplers and typically consisted of the top few inches of sediment. Water and sediments were collected along straight line traverses across rivers and harbors. At least three samples were collected on each traverse at points one fourth, one half, and three fourths of the distance across the water body. Water samples were preserved by immediately adding methylene chloride and extracting for one minute in the field. Sediments were preserved by storing at -10°F prior to extraction.

Our quality assurance effort was also extended to the sampling phase of our program (slide 13). At each site, field blanks were taken to check for contamination from the sampling apparatus. Field spikes were included to be certain our preservation techniques were adequate. Also, replicates were taken to determine the precision of our data with regard to both sampling and analytical portions of the work.

The data that we have accumulated in our monitoring studies are summarized in the next slide (slide 14). All numbers are expressed in ppb. As you can see, most of our sampling sites are located in the Midwest with the exception of San Francisco Bay. The sites were selected to be representative of industrialized environments. That is, we did not limit our sampling to some pristine mountain lake or stream. All of these bodies of water receive, either directly or indirectly, discharges from industrial and municipal sewage treatment plants. The numbers are averages of the data obtained from one or more traverses. ND stands for non-detected. The numbers in parentheses are values that we have quantitated but that are actually below our limit of reliable detection, 0.5 ppb for water and 100 ppb for sediment. Therefore, these levels may have come from our own laboratory contamination. In any case, they represent the maximum concentration which may have been present in the samples. Surveying the data, you can see that the only really significant amount of butyl benzyl phthalate was found in the sediment from Saginaw River. The water from the Saginaw River is relatively clean. This suggests that the butyl benzyl phthalate found in the sediment was discharged some time ago or is currently being discharged but in a discontinuous manner.

Overall, these results agree quite nicely with results from our laboratory environmental fate experiments. We have found that butyl benzyl phthalate undergoes rapid degradation in sewage treatment plants and in natural waters, but is degraded much more slowly under anaerobic conditions which might be present in river bottom sediments.

The concentrations of butyl benzyl phthalate present in these environmental waters provide a large margin of safety for all of the aquatic organisms that we have tested. However, the same conclusion cannot necessarily be reached for all phthalate esters. The next slide illustrates the variation in chronic toxicity for three different phthalates (slide 15). Di-2-ethylhexyl phthalate is reported to have the lowest effect level for both Daphnia and fish. Di-n-butyl is reported to be much more toxic to Daphnia than butyl benzyl but they have approximately the same effect level for fish. Therefore, even though the compounds are closely related structurally, they have significantly different toxicological properties.

Based on this evidence, we strongly urge everyone performing risk assessments to consider all compounds on an individual basis when establishing water quality criteria and standards instead of grouping them in structurally related classes. This avoids the problem of establishing unnecessarily strict standards for the entire group based on the most highly toxic or least desirable compound within the group.

In establishing water quality criteria and standards, we believe the points on the following slide form the basis of a scientifically sound assessment (slide 16). First of all, what detection limits or analytical capabilities are available on a routine or semi-routine basis? Obviously it would be difficult to enforce a water quality standard that is set below the state of the art of reliable analytical determination. Secondly, what is the toxicity or effect level of the specific compound of interest? Making judgments on structurally similar compounds may result in standards that are set too conservatively or too leniently. Finally, what concentrations are currently present in the environment? This information helps to establish a data base and determine whether the aquatic toxicities measured in the laboratory are environmentally relevant.

In summary, we have developed a sensitive analytical method for butyl benzyl phthalate under a strict program of good laboratory practices (slide 17). We have sampled waters from a number of different industrialized rivers and harbors and found the average environmental concentration to be less than 1 ppb. When viewed in the light of chronic toxicity tests for Daphnia and fathead minnows, a safety margin of approximately 1000 currently exists for butyl benzyl phthalate in the environment. We hope information like this will be carefully considered by agencies formulating water quality criteria and standards.

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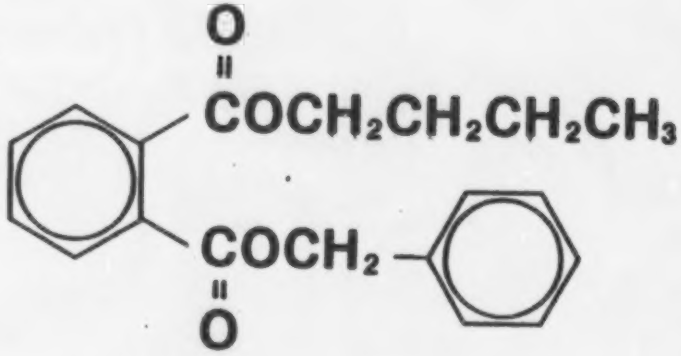
SLIDE 1

BUTYL BENZYL PHTHALATE
SANTICIZER® 160



SLIDE 2

Production Volume > 100 x 10^6 lb/yr

Uses: Plasticizer for PVC and
Other Synthetic Resins

Occurrence: Flooring Material
Household Products

SLIDE 3

ACUTE LETHALITY OF BUTYL BENZYL PHTHALATE

Species		96 Hr. LC_{50} (PPB)
Algae —	*Microcystis*	10*
	Dunaliella	1000
	Navicula	600
	Skeletonema	600
	Selenastrum	400
Invert. —	*Daphnia Magna*	3700
	Mysid Shrimp	900
Fish —	**Fathead Minnow**	5300
	Fathead Minnow	2100
	Bluegill	1700
	Rainbow Trout	3300
	Sheepshead Minnow	3000

SLIDE 4

CHRONIC TOXICITY OF BUTYL BENZYL PHTHALATE

Species	MATC (PPB)
Daphnia Magna	260-760
Fathead Minnow	140-360

SLIDE 5

AQUEOUS SAMPLES

Extract with CH_2Cl_2

↓

Dry with Na_2SO_4

↓

Concentrate

↓

Analyze

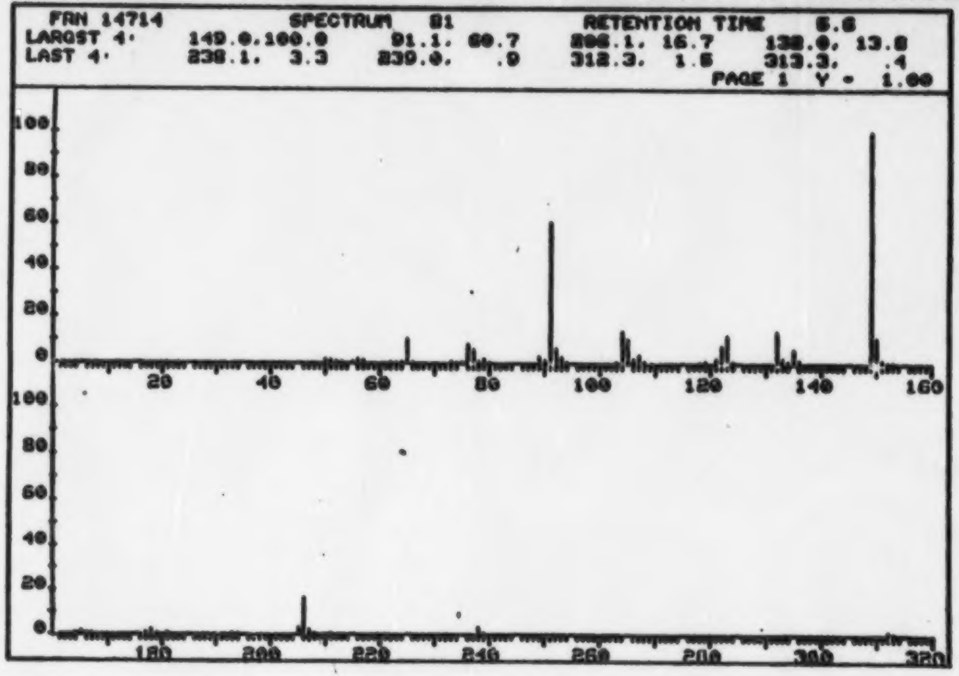
SLIDE 6

SEDIMENT SAMPLES

Add Cyclohexane
↓
Homogenize
↓
Centrifuge
↓
Concentrate
↓
Analyze

SLIDE 7

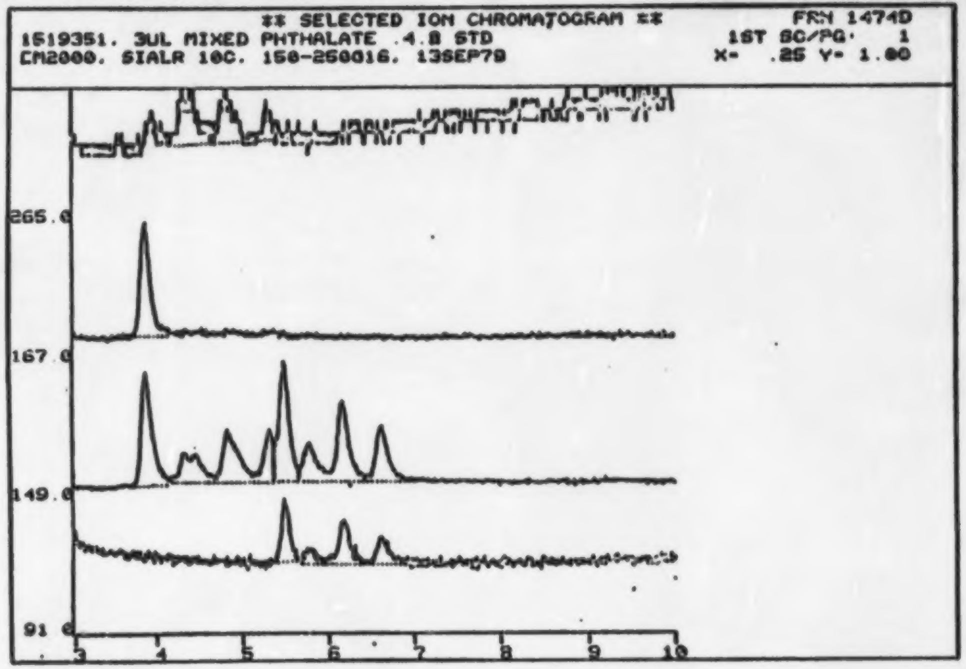
MASS SPECTRUM OF BUTYL BENZYL PHTHALATE



```
 FRN 14714            SPECTRUM   81          RETENTION TIME    5.6
LARGST 4·     149.0,100.0    91.1, 60.7    206.1, 16.7    132.0, 13.8
LAST 4·       238.1,  3.3   239.0,   .9    312.3,  1.5    313.3,   .4
                                                       PAGE 1  Y = 1.00
```

SLIDE 8

SELECTED ION MONITORING OF PHTHALATE STANDARDS



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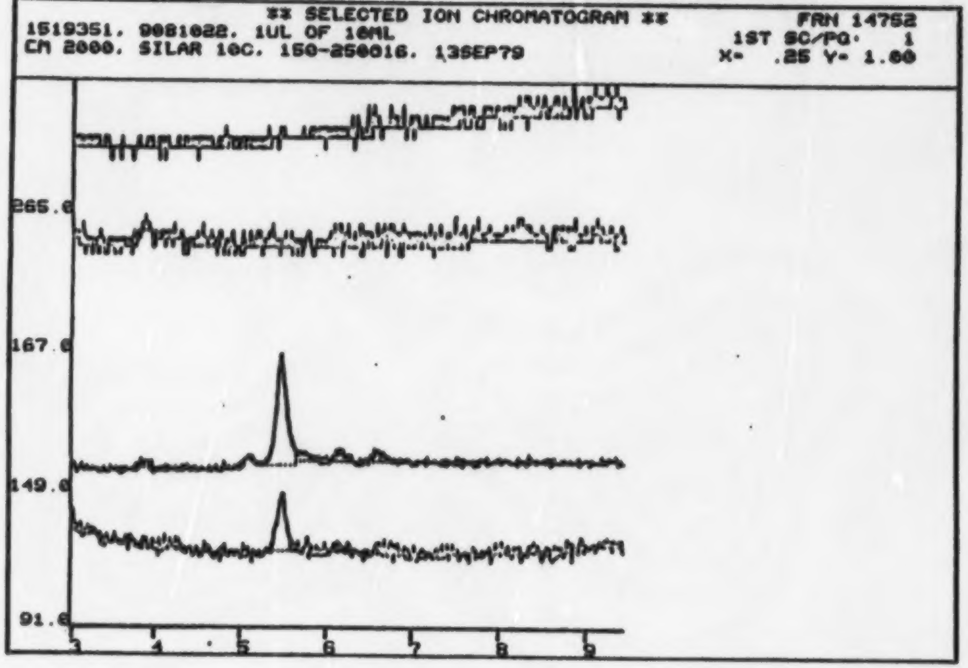
                ** SELECTED ION CHROMATOGRAM **                FRN 1474D
1519351. 3UL MIXED PHTHALATE .4.8 STD                        1ST SC/PG·    1
CM2000. SIALR 10C. 150-250016. 13SEP79                       X-   .25 Y- 1.00
```

SLIDE 9

SIM OF PLANT EFFLUENT EXTRACT



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                ** SELECTED ION CHROMATOGRAM **                  FRN 14752
1519351. 9081022. 1UL OF 10ML                                 1ST SC/PG·   1
CM 2000. SILAR 10C. 150-250016. 13SEP79                      X-   .25 Y- 1.00
```

SLIDE 10

QUALITY ASSURANCE

Method Validation

- Instrument Calibration
- Spiked Samples
- Replicates

SLIDE 11

ACCURACY AND PRECISION

	Level	Recovery	% RSD
Water	0.5 ppb	108	20
	1.0	91	36
	5.0	81	13
	10.0	80	21
Sediment	0.1 ppm	131	9.2
	0.5	96	30
	1.0	102	7.9

SLIDE 12

SAMPLING

- Water Column
- Traverse

SLIDE 13

QUALITY ASSURANCE

Sampling

- Field Blanks
- Field Spikes
- Replicates

SLIDE 14

ENVIRONMENTAL CONCENTRATIONS OF BUTYL BENZYL PHTHALATE

(PPB)

Location	Water	Sediment
Waukegan Harbor & Creek (Illinois)	(0.24)	ND
Saginaw River (Michigan)	(0.43)	480
Illinois River (Illinois)	(0.47)	ND
Meramec River (Missouri)	(0.38)	ND
Missouri River (Missouri)	(0.23)	(50)
Mississippi River (Missouri)	1.4	ND
San Francisco Bay	ND	—

SLIDE 15

VARIATION IN CHRONIC TOXICITY

	Effect Level (PPB)	
Phthalate	**Daphnia**	**Fish-MATC**
Di-n-butyl	20	90-300
Di-2-ethylhexyl	3	5-14
Butyl Benzyl	760	140-360

SLIDE 16

WATER QUALITY CRITERIA & STANDARDS

- Detection Limits
- Toxic Effects
- Current Environmental Concentrations

SLIDE 17

SUMMARY

- Sensitive Analytical Method
- Average Conc. in Water </ PPB
- Safety Margin $\approx 10^3$

PREDICTING CONCENTRATIONS OF CHEMICALS IN THE ENVIRONMENT

SECTION VI

FATE AND TRANSPORT OF TOXICS WITH RESPECT TO WATER QUALITY CRITERIA

Jerald L. Schnoor

Energy Engineering Division
Civil and Environmental Engineering
The University of Iowa
Iowa City, Iowa 52242

ABSTRACT

Fate and transport of the pesticides dieldrin, alachlor, and atrazine have been simulated in Coralville and Rathbun Reservoirs, Iowa. The soluble herbicides alachlor and atrazine have been shown to biologically degrade and to be transported out of the Reservoirs in the outflow. Their rate of degradation is significant, and rate constants range from 0.01 - 0.06 per day. The fate and transport of the neutral, hydrophobic insecticide dieldrin is considerably different.

From the analysis it was determined that ∿ 40% of the dieldrin inflow to Coralville Reservoir is lost to the bottom via sedimentation and 50% is released through the dam gates of this short detention time Reservoir. Uptake by fish accounts for about 10% of the dieldrin input due to the extremely large biomass of biota, 1000 lbs/acre. The partitioning of dieldrin in the water column is 74% in the fish, 25% dissolved in the water, and less than 1% adsorped to suspended solids. Mean residues in the edible tissue of bottom feeding fish have declined below the FDA guideline of 300 ppb. Photodegradation, volatilization, and biological degradation reactions are so slow as to be neglible in the water column compared to the transport rates, but an overall degradation rate of ∿ 0.1% per day of dieldrin in sediment contributes to the gradual decline of sediment concentrations. Fish and sediment concentrations are essentially in equilibrium with mean dissolved dieldrin concentrations.

Field monitoring and modeling of the bioconcentration factors (BCF) in several Iowa Rivers and Lake Michigan have indicated that BCF's which are normalized on a lipid basis are nearly equal to the octanol water partition coefficient. Proposed EPA water quality criteria for selected neutral, hydrophobic organics (dieldrin, DDE, chlordane, and PCB's) are discussed.

INTRODUCTION

The insecticide of widest usage for corn rootworm and cutworm during the 1960's and 1970's was the chlorinated hydrocarbon aldrin (1,8,9,10,11,11-Hexachloro-2,3-7,6-*endo*-2,1-7,8-*exo*-tetracyclo [6.2.1.$1^{3,6}$. $0^{2,7}$] dodec-4,9-diene) a derivative of hexachlorocyclopentadiene. Aldrin is rapidly converted to its epoxide dieldrin (1,8,9,10,11,11-Hexachloro-4,5-*exo*-epoxy-2,3-7,6-*endo*-2,1-7,8-*exo*-tetracyclo [6.2.1.$1^{3,6}$. $0^{2,7}$] dodec-9-ene) via biodegradation in natural water (1). Due to the widespread usage of aldrin in Iowa as a corn insecticide, Iowa waters have become contaminated with dieldrin.

During the mid-1960's aldrin usage in Iowa reached a peak application of 6.5 million pounds per year over a total of approximately five million acres. Insect resistance to this chlorinated hydrocarbon became prevalent in the mid-to-late 1960's and swept from west to east across Iowa, and usage declined from 6.5 to approximately 3.5 million pounds per year. In 1975 the United States Environmental Protection Agency (USEPA) cancelled the registration of aldrin and dieldrin insecticides.

Public attention in Iowa to the problems of aldrin and dieldrin reached a peak in 1976 when the USEPA and the Iowa Conservation Commission (ICC) placed a ban on commercial fishing in Coralville Reservoir, located in eastern Iowa. The primary commercial species affected by the ban was big-mouth buffalo, a bottom-feeding "rough" fish. In 1975, several big-mouth buffalo, catfish and carp were shown to be in excess of the United States Food and Drug Administration (FDA) guideline of 0.3 mg/kg in edible tissue. Recent analyses (1978-1980) have shown that dieldrin residues in big-mouth buffalo and other bottom feeding fish are declining and that the average composite catch is below the FDA guideline. On November 7, 1979, the ban on commercial fishing in Coralville Reservoir was lifted by action of the ICC in cooperation with the FDA. It is instructive to follow the fate of dieldrin through this chronology of peak precursor application, cancellation of the pesticide, ban on commercial fishing and lifting of the ban. A simple mathematical model was developed to analyze these trends and to make future projections.

DISCUSSION

Dieldrin Trends in Iowa Rivers

Eight major Iowa rivers (the Cedar, Iowa, Little Sioux, Mississippi, Missouri, Nishnabotna, Racoon, and Skunk Rivers) were analyzed for dieldrin concentrations in monthly grab samples of unfiltered water (2), from 1968 to present. A decreasing trend was noted in measurable dieldrin levels since 1968 in all cases. Furthermore logarithmic least squares regression analyses (log concentration vs. time) for the Cedar, Iowa, Little Sioux, Nishnabotna, Racoon, and Skunk Rivers yielded negative slopes and significant correlation coefficients at the 95% confidence level. The average decline was 10% per year or a half-life of about 7 years. Mean measurable concentrations in 1968-70 were 0.015 μg/l and in 1978 0.004 μg/l, considerably greater than the proposed water quality criteria for dieldrin of 0.0019 μg/l on a 24 hour average basis (3).

Figure 1 presents the dieldrin and flow time series for the Iowa River below Coralville Reservoir. Peak dieldrin levels in the water correspond with periods of maximum flow. Both the mean annual dieldrin concentration as well as the envelope of the peaks decreased from 1969-1978 at a rate of approximately 15% per year. A significant correlation coefficient exists between the total dieldrin concentration and turbidity for eight Iowa Rivers from 1968-1978 (r = 0.55, n = 591 samples) (4).

Water Quality Criteria

Water quality criteria have been proposed for 129 consent-decree priority pollutants by the U.S. Environmental Protection Agency (Federal Register 44 FR 15926, 43660, 45528).

Many of these pollutants are neutral, hydrophobic chemicals with significant potential for bioaccumulation. In several cases water quality criteria have been proposed based on reported literature values of the bioconcentration factors. In this paper a comparison between field data and the laboratory studies which form the basis for the proposed water quality criteria is presented for the pesticides dieldrin, DDE, chlordane, and the industrial chemical PCB. Bioconcentration factors (BCF) in fish from eight Iowa Rivers, the Mississippi River, and the Great Lakes provide a field validation of water quality criteria.

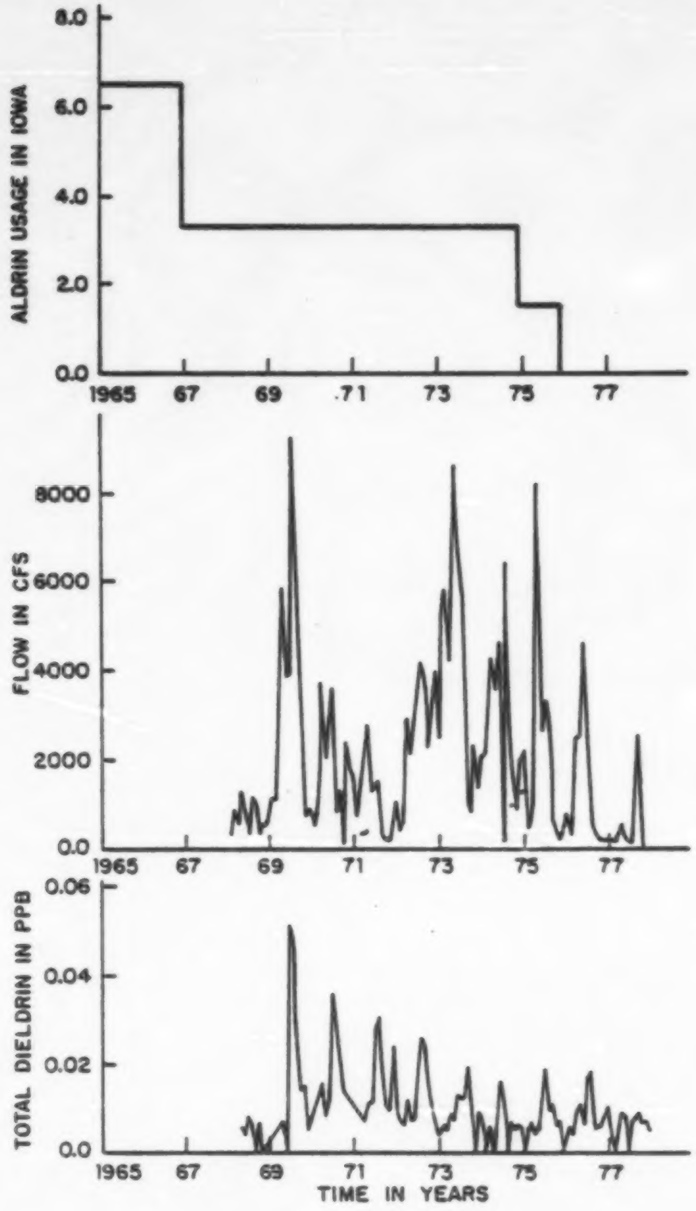


Figure 1. Estimated Aldrin Usage (in million pounds) in Iowa and Total Dieldrin Concentrations in the Iowa River at Iowa City, 1968 - 1978.

The proposed EPA 24-hour average dieldrin criteria is 0.0019 μg/l, which was determined as the allowable FDA residue level for animal feed (30 μg/kg) divided by the geometric mean BCF of seven laboratory studies (15,482 μg/kg per μg/l). (See Table 1.) Bioconcentration data from Coralville Reservoir supports the proposed criteria for piscivorous fish but indicates a more stringent criteria should be set for fatter bottom-feeding fish. Bioconcentration factors which are normalized on a lipid basis would provide a more defensible criteria for these neutral, hydrophobic pollutants. Unfortunately the required laboratory and field data simply do not exist in most cases.

Fat-normalized BCF's are independent of species and the portion of fish analyzed, and they are directly related to the octanol-water partition co-efficient (Table 2). Approximately 80% of the variance in dieldrin residues was removed by normalizing all data on a fat basis. Proposed water quality criteria for dieldrin, DDE, and PCB can be validated with field data from Iowa rivers and Lake Michigan. The proposed criterion for chlordane of 0.024 μg/l is called into question.

TABLE 2. COMPARISON OF OCTANOL/WATER PARTITION COEFFICIENTS TO BIOCONCENTRATION FACTORS ON AN OIL BASIS FROM FIELD DATA

PESTICIDE	LOG P or LOG K_{OW} Octanol/H_2O	LOG BCF (Oil Basis) Field Data
Dieldrin	4.6	5.7
DDE	5.7	5.9
Chlordane	6.0	6.2
PCB (w/5Cl)	6.0	6.2

Fate and Transport of Dieldrin in Coralville Reservoir

Coralville Reservoir is a mainstream impoundment of the Iowa River in Eastern Iowa approximately three river miles above Iowa City. It is a shallow reservoir with a mean depth of 8 feet and a mean hydraulic detention time of only 14 days. Due to large nutrient inputs from agricultural runoff, it is extremely productive in terms of primary production and fish production with a standing crop of more than 1000 lbs/acre of big-mouth buffalo alone (5). Although there exists a paucity of data, dieldrin in the sediment, dieldrin residues in bottom

TABLE 1. RELATIONSHIPS BETWEEN EXPOSURE CONCENTRATION AND RESULTING RESIDUE LEVELS IN FISH FROM FIELD STUDIES

Exposure Concentration µg/l	x	Bioconcentration BCF	=	Residue Level µg/kg	Field Location
		Dieldrin			
0.0019 Proposed 24 hr. ave.		15,482 $\overline{X}_g$, whole body		30	FDA Animal Feed
0.00044		68,286, Lake Trout Whole Body		30	FDA Animal Feed
0.0033		76,000 Bottom Feeders Edible Tissue ∿ 10% oil		250	Coralville Reservoir 1979
0.0033		$5 \times 10^5 \frac{\mu g/kg\ oil}{\mu g/l}$		$1650 \frac{\mu g\ dieldrin}{kg\ oil}$	Coralville Reservoir 1979
		DDE			
0.002		50,000 Bottom Feeders Edible Tissue ∿ 10% oil		100	Iowa R./Coralville Nodway Miss. R.

Exposure Concentration µg/l	x	Bioconcentration BCF	=	Residue Level µg/kg	Field Location
		DDE			
0.002		$8 \times 10^5 \frac{\mu g/kg\ oil}{\mu g/l}$		$1600 \frac{\mu g\ DDE}{kg\ oil}$	Iowa R./Coralville Nodway Miss. R.
		Chlordane			
0.001 (?)		160,000 Bottom Fish Edible Tissue 5-10% oil		160	Iowa R./Coralville Miss. R.
0.001 (?)		$1.6 \times 10^6 \frac{\mu g/kg\ oil}{\mu g/l}$		$1600 \frac{\mu g\ chlordane}{kg\ oil}$	Miss. R.
		ΣPCB's			
0.3		33,000 Piscivorous Whole Fish < 1% oil in edible		10,000	East Lake Michigan
0.3		$1.6 \times 10^6 \frac{\mu g/kg\ oil}{\mu g/l}$		$500,000 \frac{\mu g\ PCB}{kg\ oil}$	East Lake Michigan

Table 1. (Continued)

feeding fish, and dieldrin in the water column have all decreased roughly by a factor of 4 since 1969-70 (Figure 2). Sorption studies and field data indicate that the equilibrium partitioning between suspended sediments and water is approximately 1,000-6,000 μg of dieldrin per kg of dry sediment per μg/l of water for fine grained sediments. The bioconcentration factor or the ratio of dieldrin residues in the wet, edible tissue of bottom feeding fish compared to the whole water are ∿30,000 μg/kg fish per μg/l water. All concentrations are decreasing at roughly 15% per year.

A mathematical model was developed to simulate the fate and transport of dieldrin through Coralville Reservoir. For a completely mixed system, the following equations formulate the principle of continuity of mass in the Reservoir ecosystem. These equations expand on a prior development (6), and details of the model are presented elsewhere (7). In the water column, the rate of change of total concentration C_T in terms of the dissolved and particulate concentrations is:

$$\frac{dC_T}{dt} = \frac{W(t)}{V} - \frac{C_T}{t_o} - \sum kC - k_s C_p - k_1 C \qquad (1)$$

in which:

C_T = total dieldrin concentration in unfiltered water, L^3

t = time, T

$W(t)$ = rate of mass input of dieldrin, MT^{-1}

V = reservoir volume, L^3

t_o = mean hydraulic detention time, T

Σk = the sum of the pseudo-first order decay rate constants including biological degradation, chemical hydrolysis, direct photolysis, and volatilization, T^{-1}

C = dissolved dieldrin concentration, ML^{-3}

k_s = mean sedimentation rate constant, T^{-1}

C_p = particulate dieldrin concentration, ML^{-3}

k_1 = biouptake rate constant, $\left(\frac{\text{liters filtered}}{\text{kg fish-day}}\right)\left(\frac{\text{kg fish}}{\text{liter}}\right)$

The total concentration of dieldrin in whole water is the sum of the particulate and dissolved phases:

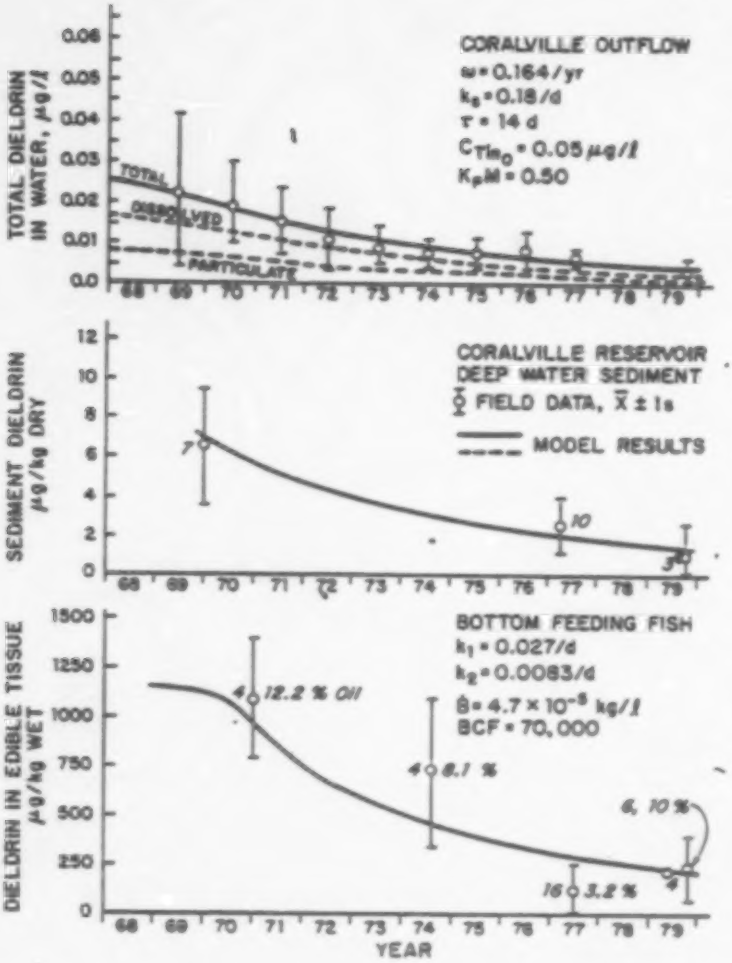


Figure 2. Field Data and Model Results for Dieldrin in Coralville Reservoir. Numbers represent the number of data points and percentages are the percent oil or lipid content of the catch.

$$C_T = C + C_p \qquad (2)$$

Adsorption and desorption rates are orders of magnitude greater than the decay and transport processes which formulate equation (1). Thus instantaneous sorptive equilibrium is achieved between the two phases very rapidly in comparison to sedimentation, washout, volatilization, photolysis, hydrolysis, biological degradation, and biological uptake. Therefore the concentrations, C and C_p may be replaced by their equivalents in terms of C_T providing that the equilibrium adsorption isotherm is known. Linear adsorption isotherms have been reported by Karickhoff (8):

$$r = K_p C \qquad (3)$$

in which

K_p = linear adsorption partition coefficient, $(M/M)/(M/L^3)$

r = amount of pesticide adsorbed per unit mass of dry sediment, M/M

It follows from equations (2) and (3) that C and C_p may be expressed in terms of C_T under conditions of sorptive equilibrium:

$$C = \frac{C_T}{(1 + K_p M)} \qquad (4)$$

$$C_p = \frac{C_T K_p M}{(1 + K_p M)} \qquad (5)$$

Substituting for C_p and C from the above relationships into equation (1) the mass balance differential equation is:

$$\frac{dC_T}{dt} = \frac{W}{V} - \frac{C_T}{t_o} - \frac{\Sigma k + k_1}{1 + K_p M} C_T - \frac{k_s K_p M}{1 + K_p M} C_T \qquad (6)$$

Bioaccumulation follows the linear kinetics of uptake minus depuration:

$$\frac{dF}{dt} = (k_1 f_1 C_T/B) - k_2 F \qquad (7)$$

in which $f_1 = \frac{C}{C_T} = \frac{1}{(1 + K_p M)}$ = fraction of dissolved dieldrin

k_2 = depuration rate constant, T^{-1}

$$k_2 = \frac{k_1 \text{ (dissolved fraction)}}{\text{(Biomass)(Fish Partition)}} = \frac{(k_1/\text{day})\left(\frac{1}{1 + K_pM}\right)}{\left(\frac{\text{kg fish}}{\ell}\right)\left(\frac{\mu g/kg}{\mu g/\ell}\right)}$$

F = whole body fish residue level, M/M wet weight

B = fish biomass concentration, M/L^3 wet weight

Equations (6) and (7) may be solved analytically for constant co-efficients and simple pesticide loading functions, W(t), or they may be integrated numerically. In the case of a pesticide ban, the W(t) might typically decline in an exponential manner due to degradation by soil organisms. For an exponentially declining loading function at rate ω, the analytical solutions to equations (6) and (7) are:

$$C_T = C_{T_o} e^{-\delta t} + \frac{C_{Tin_o}}{\varepsilon}\left(e^{-\omega t} - e^{-\delta t}\right) \tag{8}$$

$$F = F_o e^{-k_2 t} + \frac{k_1 f_1}{B}\left\{\frac{C_{T_o}}{\gamma} e^{-\delta t} + \frac{C_{Tin_o}}{\varepsilon\theta} e^{-\omega t} - \frac{C_{Tin_o}}{\varepsilon\gamma} e^{-\delta t}\right\}$$
$$- \frac{k_1 f_1}{B} e^{-k_2 t}\left\{\frac{C_{T_o}}{\gamma} + \frac{C_{Tin_o}}{\varepsilon\theta} - \frac{C_{Tin_o}}{\varepsilon\gamma}\right\} \tag{9}$$

in which:

$f_2 = \frac{C_p}{C_T} = \frac{K_pM}{(1 + K_pM)}$ = fraction of particulate pesticide

C_{T_0} = initial total pesticide concentration in lake, M/L^3

C_{Tin_o} = initial total pesticide inflow concentration, M/L^3

ω = rate of exponentially declining inflow concentration, T^{-1}

$\alpha = (\Sigma k) f_1 + k_1 f_1 + k_s f_2$, T^{-1}

$\gamma = k_2 - \alpha - (1/t_o)$, T^{-1}

$\delta = \alpha + (1/t_o)$, T^{-1}

$\varepsilon = \alpha t_o + 1 - \omega t_o$, dimensionless

$\theta = k_2 - \omega$, T^{-1}

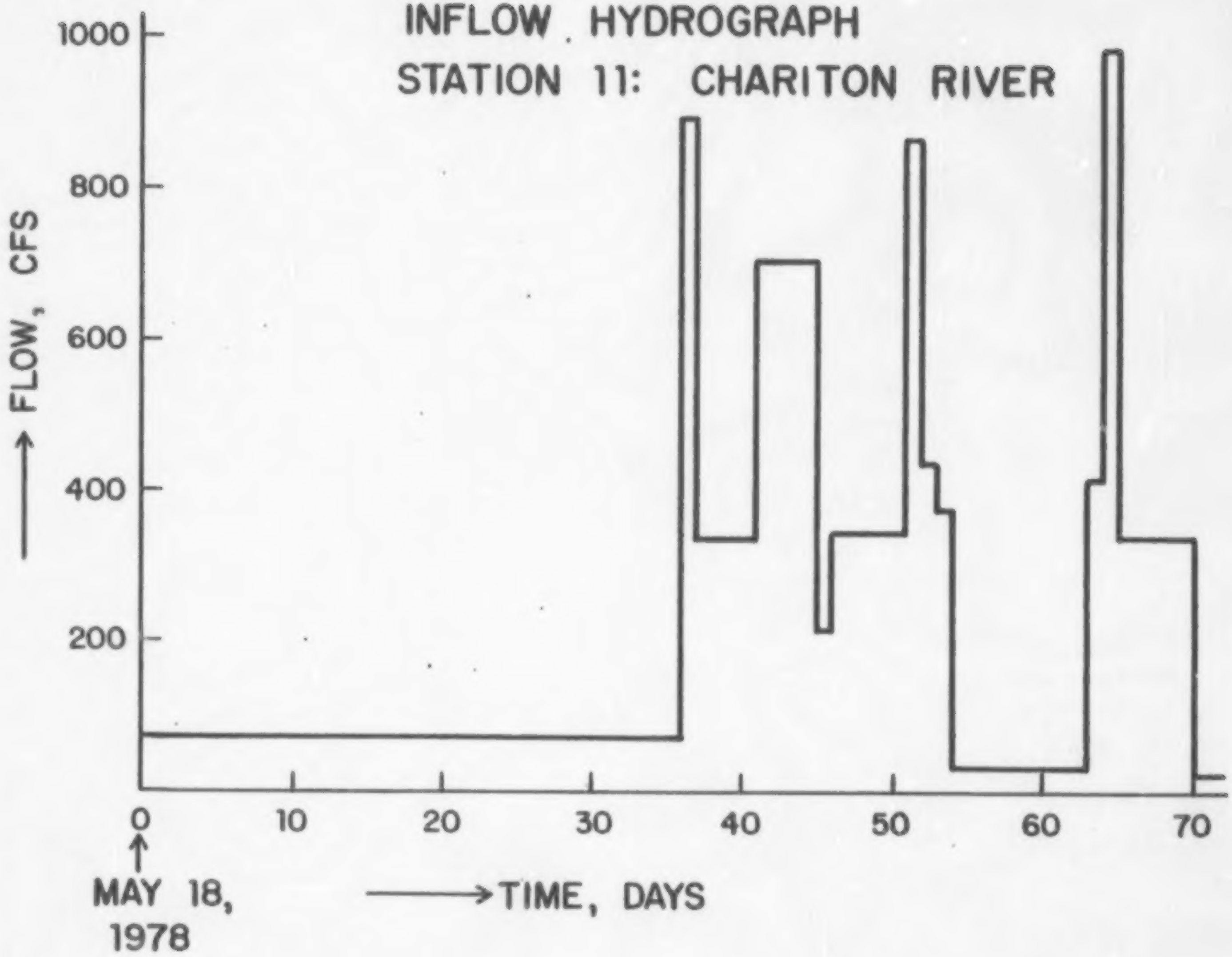
Equations (6) and (7) are written for only one water compartment with sedimentation of suspended solids into the sediment compartment. It is straightforward to extend the analysis to a number of compartments with interflow and bulk dispersive transport between compartments (9). The equations are linear and may be solved analytically or numerically.

RESULTS AND SUMMARY

Model coefficients and results are plotted in Figure 2. From the analysis it was determined that ∿ 40% of the dieldrin inflow to the Reservoir is lost to the bottom via sedimentation and 50% is released through the dam gates of this short detention time Reservoir. Uptake by fish accounts for about 10% of the dieldrin input due to the extremely large biomass of biota, 1000 lbs/acre. The partitioning of dieldrin in the water column is 74% in the fish, 25% dissolved in the water, and less than 1% adsorped to suspended solids. Mean residues in the edible tissue of bottom feeding fish have declined below the FDA guideline of 300 ppb. Photodegradation, volatilization, and biological degradation reactions are so slow as to be negligible in the water column compared to the transport rates, but an overall degradation rate of ∿ 0.1% per day of dieldrin in sediment accounts for the gradual decline of sediment concentrations. Fish and sediment concentrations are essentially in equilibrium with mean dissolved dieldrin concentrations.

Similar modeling efforts on Coralville and Rathbun Reservoirs in Iowa for the herbicides alachlor and atrazine are quite different than dieldrin. Being quite soluble, these herbicides are shown to undergo negligible sedimentation but to biologically degrade and to be transported out of the Reservoirs via the outflow. Figures 3-6 are the results for alachlor (Lasso®) in Rathbun Reservoir. Time-variable loadings and flow were required to accurately calibrate the model to in-situ and outflow concentration data. The rate of degradation was significant, and pseudo-first order rate constants ranged from 0.01 - 0.06 per day, in relative agreement with laboratory protocol measurements of biodegradation performed at Athens EPA Laboratory.

Figure 3



INFLOW HYDROGRAPH
STATION 11: CHARITON RIVER

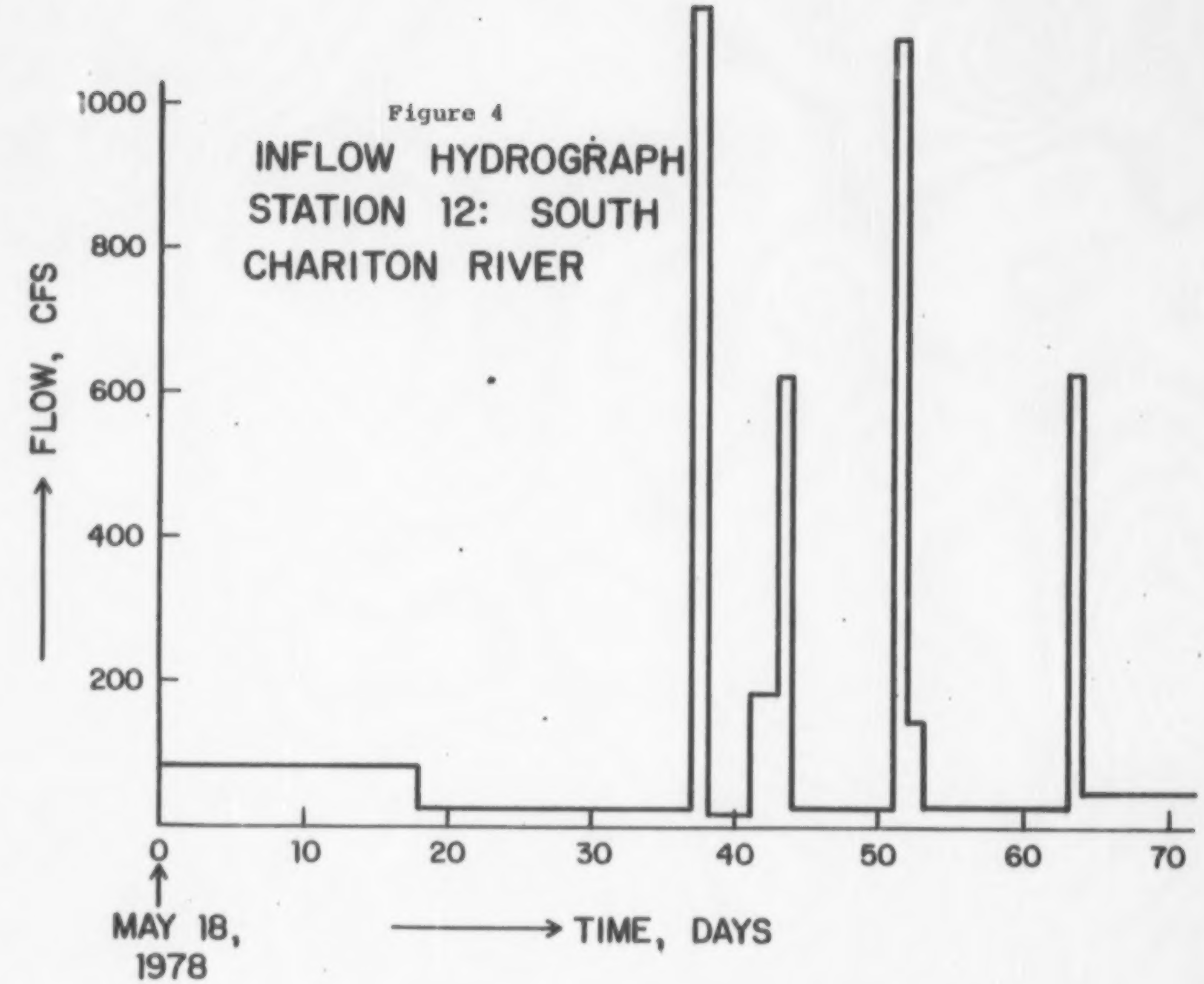


Figure 4

INFLOW HYDROGRAPH
STATION 12: SOUTH
CHARITON RIVER

Figure 5. LASSO IN LAKE RATHBUN: INPUT CONCENTRATIONS

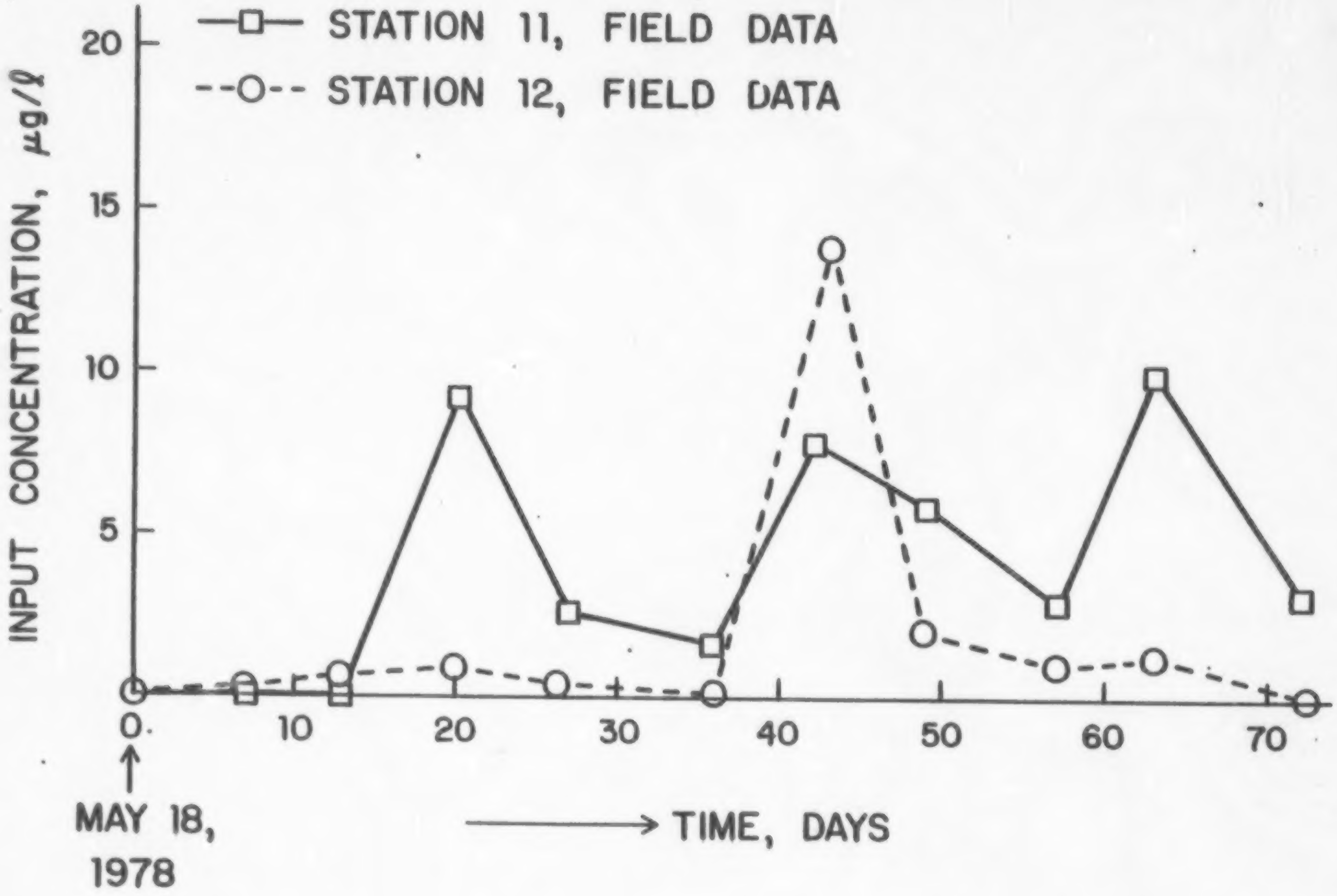
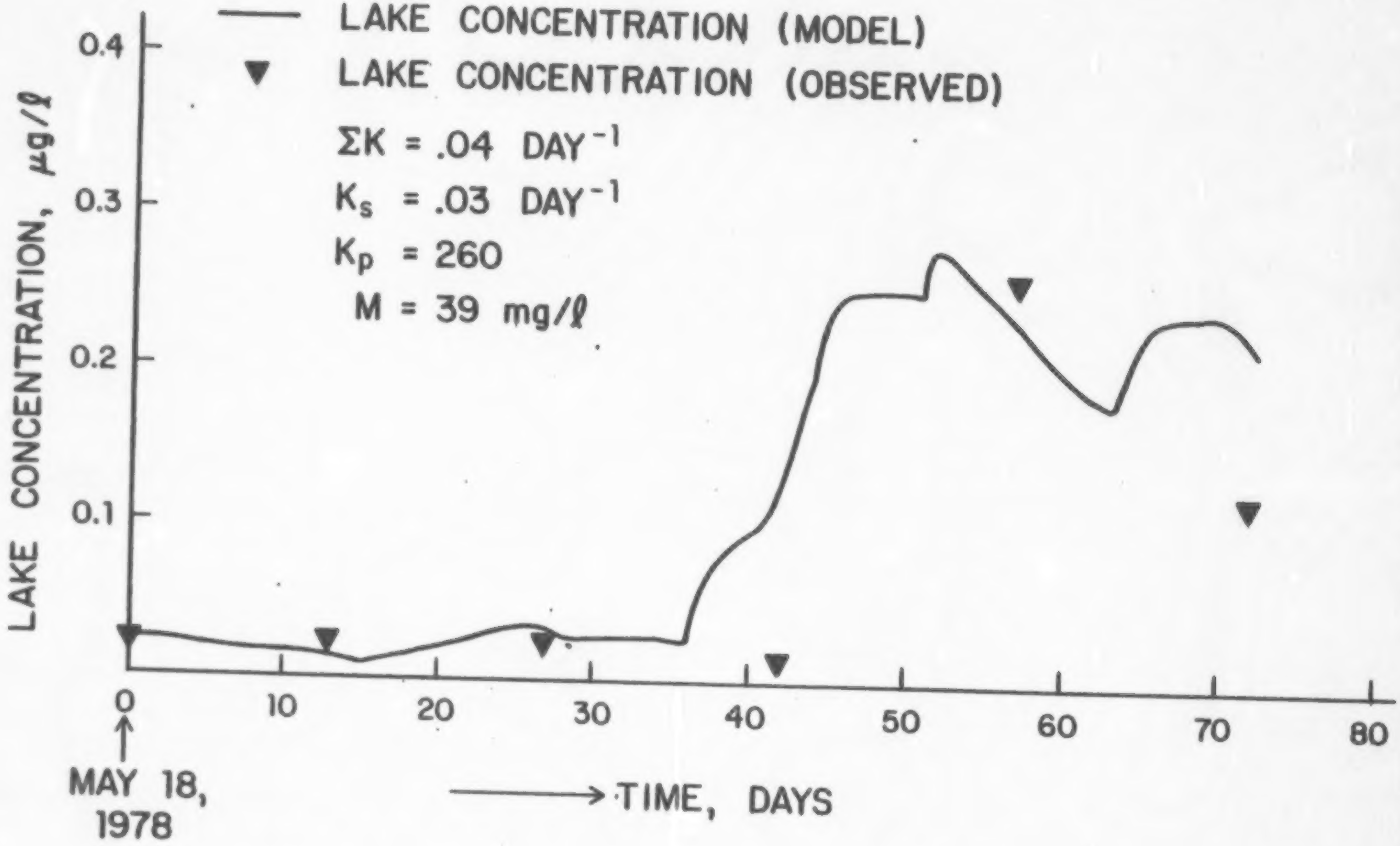


Figure 6. LASSO IN LAKE RATHBUN



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2. Samples were analyzed by L.G. Johnson, University of Iowa Hygienic Lab using USEPA procedures, a Varian gas chromatograph with electron capture detector. Samples were collected at water supply intake points.

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9. Equations 1-6 can be written for a number of water compartments which are connected by flow and suspended sediment exchanges. Therefore it is not necessary to assume completely mixed conditions for the entire Reservoir, only for a given control volume. One, two and five interconnected water compartments have been performed in this analysis. Results are shown for a two compartment model of water and sediment with exchanges in Figure 2. However, the addition of water and sediment compartments results in very minor changes to the calculations of Figure 2.

ACKNOWLEDGEMENTS

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SOME CALIBRATIONS OF CHEMICAL FATE MODELS TO FIELD DATA: A TEST OF MODEL CREDIBILITY

Dr. Robert V. Thomann

Environmental Engineering and Science Program
Manhattan College
Bronx, N.Y. 10471

ABSTRACT

The need for model credibility in the ultimate use of chemical models for allocation analysis is reviewed. The issue of model credibility is considered vitally important to the process of setting water quality standards and allowable chemical concentrations in various levels of the ecosystem. The present state of the credibility of chemical fate models is summarized and it is indicated that, in many instances, analysis frameworks have not been tested against field data.

The basic modeling framework for chemical fate is given in the form of a system of equations of the dissolved and particulate forms. The importance of including spatial distributions and chemical transport due to flow and dispersion is stressed. A simple model is given of the fate of chemicals in the aquatic food chain.

Examples are given of the calibration of the fate model to: 1) PCB distribution in the Hudson River and estuary system, 2) Plutonium 239 and PCB in the Great Lakes system, and 3)cadmium in Saginaw Bay, Michigan. Each of the principle components of the model is evaluated including partitioning of chemical onto solids, sediment water exchange due to resuspension and vertical settling of particulates. The variability of the data, uncertainty in modeling particulate transport and the importance of the active sediment in resuspension processes all contribute to difficulties in calibration of the models to field data. It is shown that response times for systems such as the Great Lakes may vary by an order of magnitude depending on assumptions of the depth of sediment available for active resuspension.

It is concluded that the relative ease with which the equations can be written belies the difficulty of calibration to actual field data.

INTRODUCTION

The modeling of the fate of toxic substances in the aquatic environment is proceeding at a rapid rate stimulated in large measure by the pressure to produce "answers" to sometimes highly emotion-charged problem settings. In a relatively short space of several years, a basic set of equations for modeling the fate of toxic chemicals has been developed with special emphasis on the interaction between the sediment, suspended solids and water. Such emphasis has been the result of the contemporary focus on highly water insoluble substances such as PCBs. The modeling equations have also incorporated the uptake and transfer in the aquatic food chain where, of course, the principal effects of toxic discharges are most relevant and publicly of concern. A set of equations is now at hand and various extensions of the equations to include additional mechanisms are being added continuously. The difficulty with the present state of affairs is that the intense and rapid pace tends to preclude an orderly development of a defensible and credible model framework that includes not only the set of equations but also a calibration and verification analysis of real field data. All too often, only certain mechanisms are compared to laboratory data with the tacit understanding that such favorable comparisons indicate a modeling framework that would be suitable for field conditions.

The simple purpose of this paper is to reiterate the need for a balance between the development of systems of model equations incorporating known mechanisms of toxic interactions with the equally important evaluation of the predictive capability of such equations when real field data are utilized. Three examples are summarized to illustrate the need for this balance: cadmium in Saginaw Bay, Michigan, radionuclides in the Great Lakes and PCBs in the Hudson estuary food chain.

BASIC EQUATIONS

The model equations for calculating the fate of a toxic substance in the aquatic environment can be conveniently divided into two categories: the physio-chemical setting and the aquatic food chain or web. The equations for the physio-chemical setting are further divided into the mass balance equations for: a) the toxicant in an operationally defined dissolved state both in the water column and in the interstitial water of the sediment, b) the toxicant sorbed on the particulate suspended material and bed sediment solids and, c) suspended solids and bed sediment solids.

The toxicant equations can be written as follows: (a definition of terms is included in the Appendix). For the dissolved form, a mass balance around a finitely defined volume, i, is

$$V_i \frac{d\phi\, c_{Di}}{dt} = \sum_j Q_{ji}\, c_{Di} + \sum_j E'_{ji}\,(c_{Dj} - c_{Di})$$

$$-\lambda_i\, \phi_i\, V_i\, c_{Di} - k_{ui}\, M_i\, \phi_i\, V_i\, c_{Di}$$

$$+ K_i\, V_i\, c_{pi} + W_{Di} \qquad (1)$$

This equation applies to the dissolved state at any location in the system, i.e. water column or interstitial water in the sediment. The first two terms represent physical transport due to advection by flow Q and dispersion due to mixing, exchange, or molecular diffusion given by E. The third term represents the loss of dissolved toxicant due to mechanisms such as evaporation, bacterial degradation and photolysis. The fourth term represents the sorption of the dissolved form to the particulate form at the rate given by k_u; the fifth term is the source of dissolved toxicant due to desorption from the particulate phase, c_p at the rate K. The last term is the external input of dissolved toxicant to the i^{th} segment.

The equation for the particulate form is:

$$V_i \frac{d\, c_{pi}}{dt} = \sum_j Q_{p,ji}\, c_{pi} + \sum_j E'_{ji}\,(c_{pj} - c_{pi})$$

$$+ k_{ui}\, M_i\, \phi_i\, V_i\, c_{Di} - K_i\, V_i\, c_{pi} + W_{pi} \qquad (2)$$

This equation applies also to both the particulate form in the water column or in the sediments. Note that the settling of particulate toxicant to the bed and resuspension from the bed to the water column are included in the first term of the equation. The third and fourth terms are the sorption and desorption terms, respectively. Eqs. (1) and (2) are kinetic equations that are coupled to a mass balance equation for the solids as given by:

$$V_i \frac{dM_i}{dt} = \sum_j Q_{ji}\, M_i + \sum_j E'_{ji}\,(M_j - M_i) \qquad (3)$$

Again for this equation, any settling or resuspension of solids is included in the first term. For example, for a water column segment, the transport due to settling and that due to resuspension would be given by:

$$-v_{sij}\ A_{ij}\ M_i + v_{uji}\ A_{ij}\ M_j$$

where the first term is the settling from i to j at velocity $v_{s,ij}$ and the second term is the resuspension flux from j to i at resuspension velocity $v_{u,ji}$.

The difficulty with Eqs. (1) and (2) is that the reactions of sorption and desorption may be fast relative to other time constants in the problem so that a large amount of computation time is needed to solve the full kinetic equations. A major improvement in solution technique can be obtained by assuming local equilibrium conditions, i.e. that the reactions are so fast that the proportion of the toxicant in the dissolved and particulate phases is fixed. A single equation for the total toxicant in each segment can then be divided and the particulate and dissolved phases can be computed from the total. For example, if the subscript 1 designates a water column segment and subscript 2 designates a sediment segment, then the following equation can be divided for the total toxicant in the water column:

$$V_1 \frac{d\,c_{T1}}{dt} = -Q\ c_{T1} + E'_{21}\ c_{T2} - E'_{12}\ c_{T1} - K_{11}\ V_1\ c_{T1} + W_{T1} \qquad (4)$$

where $$E'_{12} = E'_{D1} \frac{fD_2}{\phi_2} + v_u\ A_{12}\ f_{p2} \qquad (4a)$$

$$E'_{12} = E'_{D12} \frac{f_{D1}}{\phi_1} \qquad (4b)$$

$$K_{11} = \lambda_1\ f_{D1} + \frac{v_s\ A_{12}}{V_1} f_{p1} \qquad (4c)$$

The quantities f_p and f_D represent the fraction in the particulate and dissolved phases respectively and are given by:

$$f_p = \frac{(\pi/\phi)\ M}{(1+\frac{\pi}{\phi}\ M)} \qquad (5)$$

This latter step is, of course, a major assumption and implies complete reversibility of the sorption-desorption phenomena which may not be the case for a given toxicant. In addition, there is a considerable uncertainty about the settling and resuspension velocities that should be used for a particular water body. Since the sediments may act as a reservoir for certain chemicals that are highly sorbed by particulates, the specification of these velocities and exchange mechanisms with the bed sediment becomes particularly important.

The above brief outline of equations indicates the general framework that is presently used in most toxic substances modeling. The utility of the model equations rests largely on the ability of the system to adequately be calibrated against existing data. Several illustrations follow that will indicate some of the difficulties in this most important step of developing a modeling framework.

SOME ILLUSTRATIONS

Cadmium in Saginaw Bay

The initial compartment model for Saginaw Bay includes the five segment water transport and dispersion model developed by Richardson (1975) (Figure 1). Physical transport such as flows and exchange rates between segments were estimated for 1976 by tracing the chloride concentration through the system until computed results fit the observed chlorides concentrations. Water inputs are from the Saginaw River and Lake Huron, with the circulation generally in a counter-clockwise direction. During specific periods in the year, there is a rapid in-out movement of water from the Bay with an almost total stagnation of water in Segment 2. This type of occurrence tends to keep the solids from increasing due to inputs from other segments.

Loading inputs such as suspended solids (lbs/day) from Saginaw River (segment 1) were obtained from US EPA STORET data for 1976. The suspended solids loading reached a maximum value in March of approximately 6,300,000 lbs/day. The solids then rapidly decrease by June to about an average load of 200,000 lbs/day. Cadmium loading data were obtained from the work performed at EPA Grosse Ile Large Lakes Research station. The total cadmium loading was measured in three different segments, with Segment 1 showing the highest measured value (244 lbs/day) in March. In terms of model input and for preliminary calibration, the total cadmium was divided equally between dissolved and particulate phases of cadmium.

Various settling velocities were used in the first calibration of the cadmium Saginaw Bay model. It was concluded, however, that because of the fluctuating circulation pattern on the bay, a small settling velocity (approx. .2 m/day) would allow concentrations of solids to remain too high during periods of zero or low flow through particular segments. At a higher settling velocity, solids concentration in one segment would not be greatly affected by high concentrations in another, but by how much resuspension (lbs/day) is inputted from the bottom. In this case, the high settling velocity (1.8 m/day) would control the effect of the high concentration of solids coming from other segments (example: Segment 1 into Segment 3).

The initial partitioning between particulate and dissolved cadmium was computed by a previously calibrated one segment Saginaw Bay model. However, due to the new segmentation of five segments, the data base was redistributed to sample location vs. segment then averaged. The original partition coefficient under the new model did not respond well during attempted model calibration. Therefore, a new partition coefficient was computed for each segment using the 1976 data base.

Figure 2 shows the computed ratio of particulate to dissolved cadmium and average concentration of solids (gm/l) for each segment. Values for a' indicate a ratio between 1 to 2 (μg/ℓ P/ μg/ℓ D) for each of the first four segments. The larger difference for a' in segment 5 seems to show that higher values of particulates are found in segment 5 relative to dissolved cadmium. A potential explanation for this phenomena is the possible flushing action caused by the large flows entering segment 5 from both segments 4 and 3. This flushing of water tends to decrease the concentrations of particulate cadmium, while also significantly diluting the concentration of dissolved cadmium in segment 5.

Figure 3 shows the estimated partition coefficients for each segment. The highest values occur at the segments that exchange with Lake Huron (segments 4 and 5), while the smallest value occurs in the two smallest segments in the bay (segment 1 and 3). The high partition coefficients are caused primarily from the low concentrations of solids (m) found in segment 4 and 5. This is caused by the depth of the segments, the kind of residuals found in Lake Huron (fine particles) and the large water flow. Segments 1 and 3 in turn are shallow causing a greater change for exchange with the sediment layers plus both segments are affected by the large loads of residuals (clays) being inputted from the Saginaw River.

Table 1 is a summary of system parameters and loading inputs used for the preliminary calibration Saginaw Bay cadmium model.

Preliminary calibration of the Saginaw Bay cadmium model is shown in Figures 4 and 5. Since the external input loading and boundary conditions were not high enough to effect model calibration due to the high settling rate, an external input source was required. An internal loading input (lbs/day) representing sediment resuspension is therefore necessary. The results indicated that the largest fraction of the solids concentration is due primarily to sediment resuspension. The fine tuning of the solids system in the model was done to check if a corresponding effect was shown in the cadmium system. A value of .4 μg Cd/gm of mud resuspended was used as the loading condition for cadmium. The results, as shown in Figure 5, duplicated the approximate spatial gradient of the concentration of total cadmium in each segment but were not satisfactory in calibrating the dynamic behavior of the reported concentrations. At best, the model and the present stage of development is credible only at the order of magnitude, spatial gradients level and not at the week to week level of dynamic behavior.

TABLE 1 SYSTEM PARAMETERS AND LOADS

SYSTEM PARAMETERS

Segment	Partition Coefficient $(\frac{\mu g/gm}{\mu g/\ell})$
1	77.
2	173.
3	63.
4	615.
5	890.

LOADS (lbs/day) SOLIDS Average Value (for Year 1976)

Segment	External	Internal
1	1,680,000	14,000,000
2	—	19,750,000
3	—	10,675,000
4	—	3,220,000
5	—	5,400,000

LOADS (lbs/day) TOTAL CADMIUM

Segment	External	Internal
1	12.5	5.0
2	—	7.9
3	—	4.3
4	3.0	1.3
5	3.1	2.2

Radionuclides in the Great Lakes

A second example is offered by the use of the available radionuclide data base for the Great Lakes to test the credibility of the basic model equations 1-4. The Great Lakes data base on radionuclides affords a particularly good opportunity since the input loads are well known (in contrast to the toxic substances situation) and the data base is rather extensive. As a consequence, the plutonium-239 and cesium-137 have been used as calibrating variables (Thomann and DiToro, 1980).

A seventeen segment model of the five Great Lakes with three sediment segments under each water column segment has been used as the basic physical segmentation. Estimates of the plutonium input are primarily through atmospheric loading and extend from the early 1950's through the late 1970's. Figure 6 shows the calibration of a twenty year calculation for the lakes with initial conditions at zero everywhere and loading the lakes with the known atmospheric plutonium input. As can be seen, the water column plutonium concentration varied significantly during this loading period and the calibration is reasonably good for the latter few years of the calculation. The final test, however, really is in the calibration to the sediment data since it is in the sediments that the entire time history of the load input is stored. Figure 7 shows the comparison of calculated sediment concentration in Lake Michigan to the observed concentrations that are seen. The calibration is only marginal partly due to the crudeness of the vertical sediment segmentation.

Although the calibration is approximately reasonable in the water column, much additional research, therefore, needs to be done on the calibration to the observed sediment profile. However, one can test the sensitivity of the response times in the lakes to varying sediment depths available for resuspension. Thomann and DiToro (1980) have shown the substantial sensitivity of the response time of plutonium-239 in several of the lakes as a function of the depth of bed sediment available for resuspension. For example, if Lake Huron acts as a sink and there is no sediment resuspension, the response time due to an instantaneous drop in the plutonium load is 3.2 years. However, with a two centimeter surface bed sediment available for resuspension and exchange, the time to reach 90% of the initial value extends out 75 years. The results of the plutonium calculation for the Great Lakes indicate again the need to carefully calibrate and verify the model to existing data so as to obtain a much better understanding of the sensitivity of the solutions to key parameters. The results of the Great Lakes work indicates that the depth of the sediment that is considered to be available for resuspension and exchange with the overlying water is one of the most key parameters that influences the dynamic behavior of water column toxicants. As such, even though the equations are reasonable from a mechanistic point of view, the specification of the sediment water interaction and behavior assumes a principal role in predicting the responses of bodies of water to changes in external toxicant loadings.

PCB Fate in the Hudson River Food Chain

A final example is given from analysis of PCBs in the food chain of the Hudson River (Thomann, Pacquin and Fitzpatrick, 1980). Concentrations in the water of the Hudson estuary and river have resulted in levels of PCB in fish, specifically the striped bass that have exceeded U.S. Food and Drug Administration (FDA) guidelines. The high concentrations in the striped bass have resulted in the closing of that fishery to commercial use.

The physical-chemical sector of the model is a simplified steady state mass balance model of the water column forced by the observed PCB sediment concentrations. The model is calibrated to PCB data in the river above Troy below the site of the previous input of PCBs from the GE facilities. The output from the river model is then inputted to the lower estuary portion of the Hudson including the distribution in New York Harbor. The calculations for the Hudson estuary together with analysis of data for the period 1977-1979 (collected by USGS) permit an estimation of the dissolved PCB in the water column that is considered available for uptake by the fish. This is the emphasis of this example.

For the striped bass, the spring 1978 data (obtained by NYSDEC, Albany, N.Y.) on total PCB filet concentrations (μg PCB/g, wet weight) for the lower estuary are given approximately as follows:

	Total PCB μg/g
Median (50% of fish ≤,≥	10
10% (10% of fish ≤)	5
90% (10% of fish ≥)	30

About 40% of this total is Aroclor 1254. For the New York Harbor-Long Island region, the range of 1254 PCB reported for striped bass is less than 1 μg/g to 7 μg/g. There is no apparent gradient in 1254 PCB concentrations in striped bass between the estuary region and the harbor-western Long Island region. The PCB concentration in striped bass did not appear to be significantly correlated with lipid content. For the estuary region, the data support an approximate 10 fold increase in food chain PCB concentration, from phytoplankton of about 1-2 μg/g to striped bass of 10-20 μg/g.

For the biological sector of the model a seven compartment steady state food web model over six spatial regions of the estuary and near-shore areas provides for first estimates of PCB fluxes and concentrations. The output from the food web model is input to the age-dependent striped bass model.

The basic equation for the accumulation of PCB by the striped bass is given by:

$$\frac{d\nu}{da} = k_{uf}c - K_e + \alpha \sum_i C_i \nu_i - \frac{1}{w}(dw/da)\nu \qquad (6)$$

where ν is the PCB concentration in the striped bass (μg PCB/g(w)), k_{uf} is the uptake of dissolved PCB directly from the water (ℓ/day-g(w)), c is the dissolved water concentration (μg/ℓ), K_e represents the excretion rate of PCB (1/day), α is the assimilation efficiency (μg PCB/μg PCB) of ingested food ν_i at a feeding rate C_i (g/g-day), w is the weight (g(w)) of the striped bass and a is the age of the fish.

A simple model for the rate of change of body weight is obtained from data on weight and age of the striped bass. Thus,

$$\frac{dw}{da} = (\Sigma\, a_i' C_i - r)w \qquad (7)$$

where a_i' is the net biomass conversion efficiency and r represents weight loss due to respiration, swimming and other activities.

The approximate range of the observed striped bass PCB concentration is calibrated under a range of food concentrations and excretion rates. Figure 8 shows the results. The calibration could only be approximated to the observed data because of the wide range of reported values. As shown in Figure 8, the principal uncertainty is in the food concentration and excretion rates of PCB from the striped bass. Maximum accumulation occurs for the case of increased food PCB concentration and zero excretion. Relatively wide oscillations in striped bass PCB concentrations (e.g. range of 10-15 μg/g) are calculated as a result of migration between regions of high and low PCB concentration. For the median case, the model indicates that about 10-20% of the observed concentration is due to uptake from water only, the remainder results from food intake. The model also indicates the importance of recognizing the weight gain of the striped bass in the PCB concentration time history. For resident striped bass, the PCB concentration is estimated to be about 2.5 times that of migrating striped bass, and reaches computed levels of about 25 μg/g.

Significant uncertainties exist however in the analysis. Such uncertainties include the relationship between whole body concentration and filet concentration, the lack of PCB water data in the New York Harbor, New York Bight and Long Island Sound areas and the uncertain range of the PCB concentration in the food of the juvenile and adult striped bass. As noted, the results are particularly sensitive to the assumed level of excretion of PCBs in the striped bass. In addition, the expected response time of the water column PCB concentration in the estuary is still unknown. This particular uncertainty is centered principally in the mechanisms of suspended sediment transport and bed sediment-water column interactions related to adsorption of PCBs to the particulate matter.

DISCUSSIONS AND CONCLUSIONS

The preceeding examples have attempted to indicate, in a very general way, some of the difficulties associated with applying a set of seemingly realistic equations to real data. These comparisons indicate the importance of carefully including the interaction between the sediment and the overlying water and the interaction of various biological components with the chemical in the abiotic state. It is also indicated that the equations by themselves do not indicate the most sensitive parameters to the final calculations. The examples shown in this paper clearly indicate the marked sensitivity of the results to adsorption/desorption phenomenon, the depth of bed sediment available for resuspension and the uptake assimilation efficiency in excretion rates of the toxicant by the top predators in the food chain.

The basic purposes of modeling the fate of chemicals in the physical, chemical and biological environment are twofold: 1)to predict the fate of the chemical before manufacture, and 2) to assist in the allocation of chemicals to meet specific chemical water quality standards in receiving waters.

These uses therefore indicate a strong need to develop modeling frameworks that extend beyond the development of equations in the abstract and attempt to apply those equations to real world situations. It is only after calibration and testing of the basic modeling equations that a fully credible and defensible modeling framework is at hand. Three points can then be made: 1) models must be credible in order to be used effectively for predicting chemical fate in an allocation analysis, 2) the credibility of such chemical fate models must be established through a "track" record of calibrations and verifications to observed field data and 3) the present "track" record of contemporary chemicals is relatively poor and must be developed much further. The development of complete calibrated and verified chemical fate models results in several additional research needs. Such research needs include: 1) evaluation of the interaction of sediment through resuspension and interstitial water diffusion with the overlying water column, 2) reversibility of chemicals from particulates to the dissolved state, 3) food chain and biological interactions and potential routes of chemicals to key target organisms in the food web.

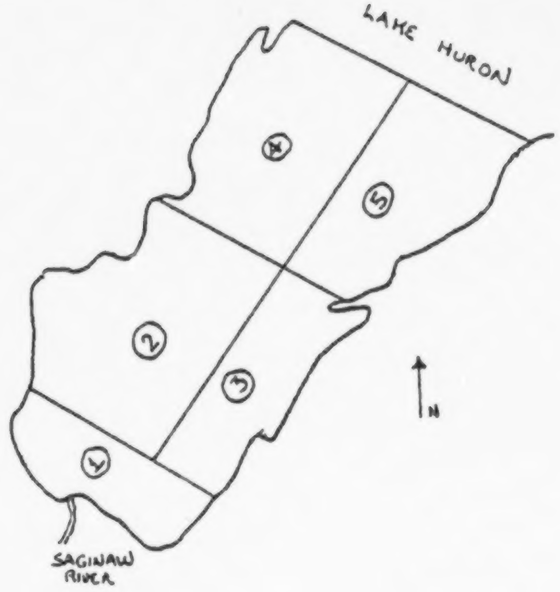


Figure 1. Saginaw Bay Segmentation

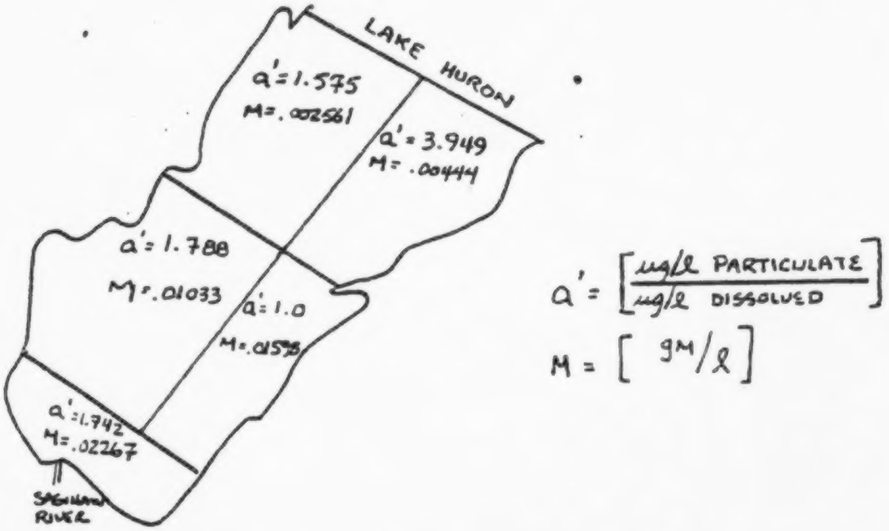
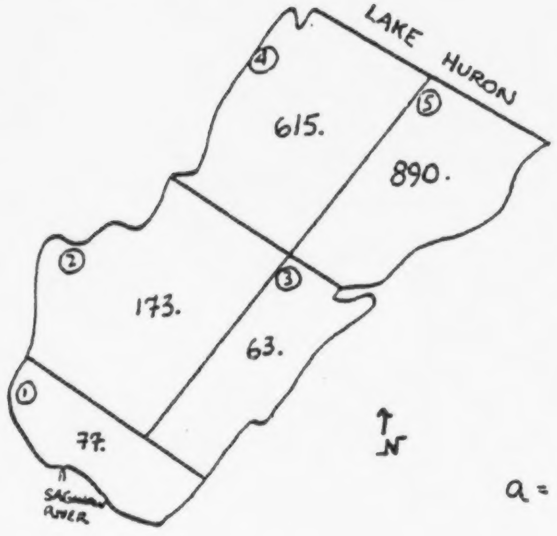


Figure 2. Ratio of Particulate to Dissolved Cadmium (a') and Average Concentration of Solids (gm/ℓ)



$$a = a'/M = \left[\mu g/gm / \mu g/l\right]$$

Figure 3. Computed Partition Coefficients for Saginaw Bay

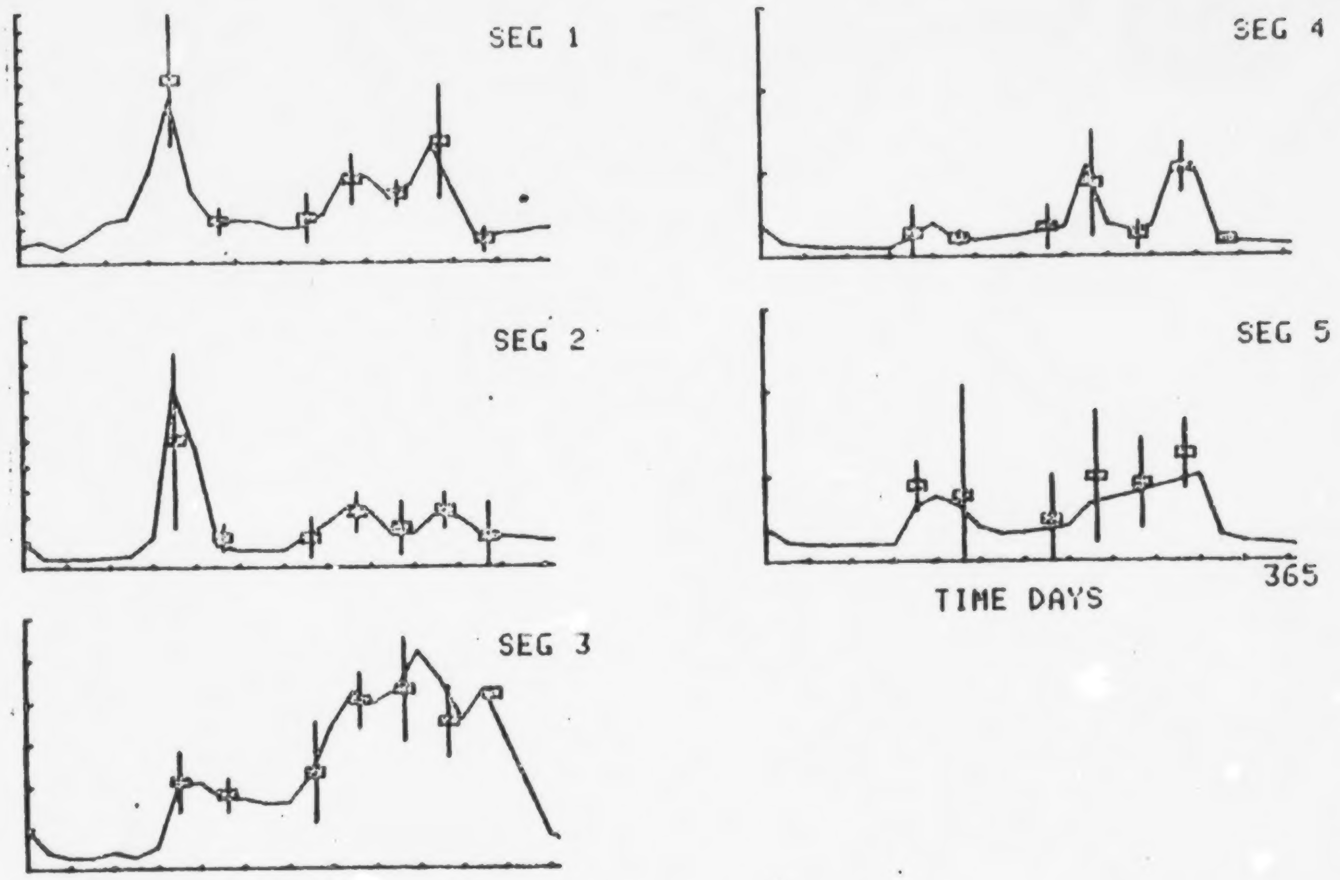


Figure 4. Solids Concentration Saginaw Bay 1976 (MG/L) DIV=5 MG/L

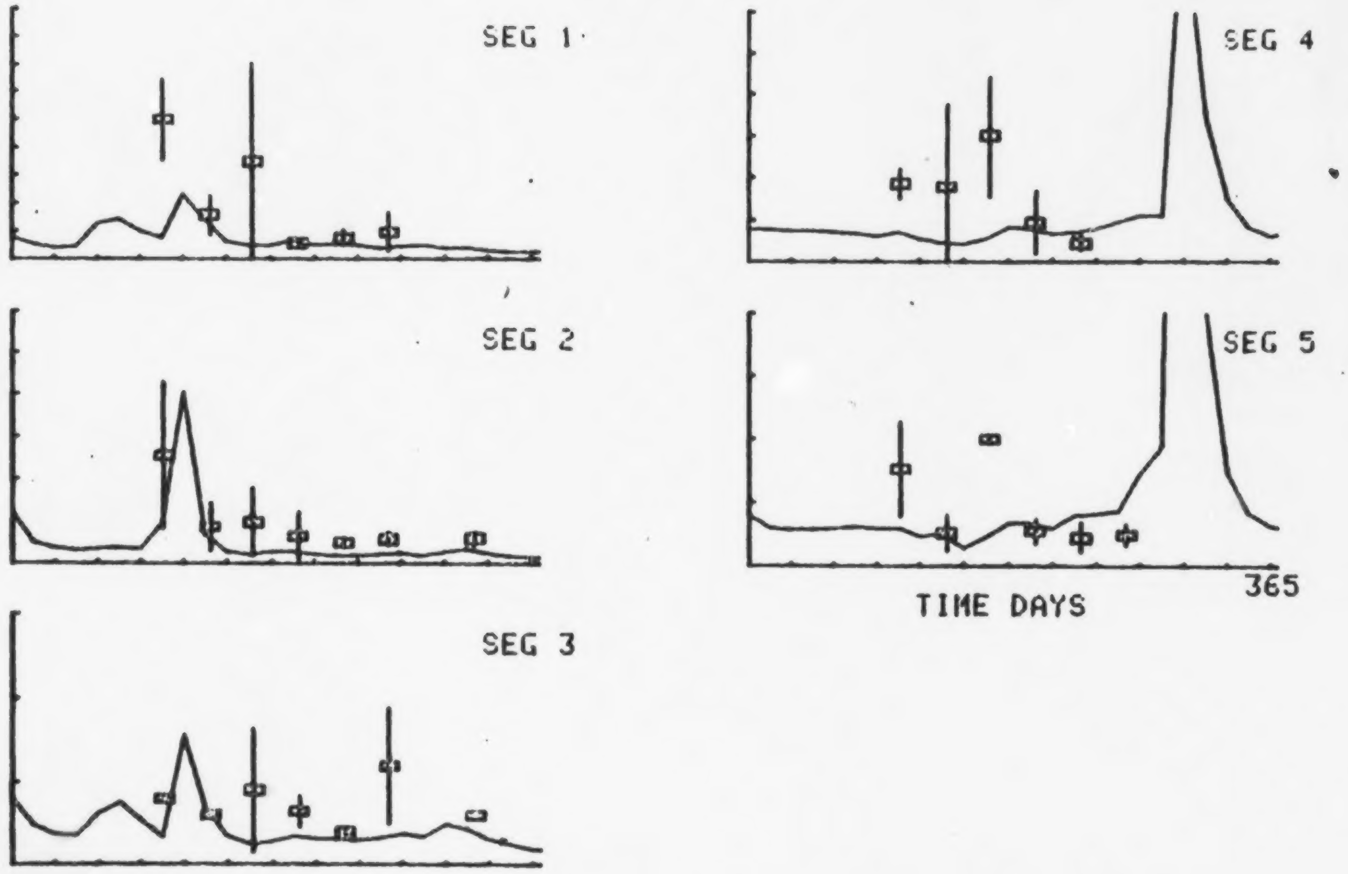


Figure 5. Total CD Saginaw Bay 1976 (µG/L) DIV=.05 µG/L

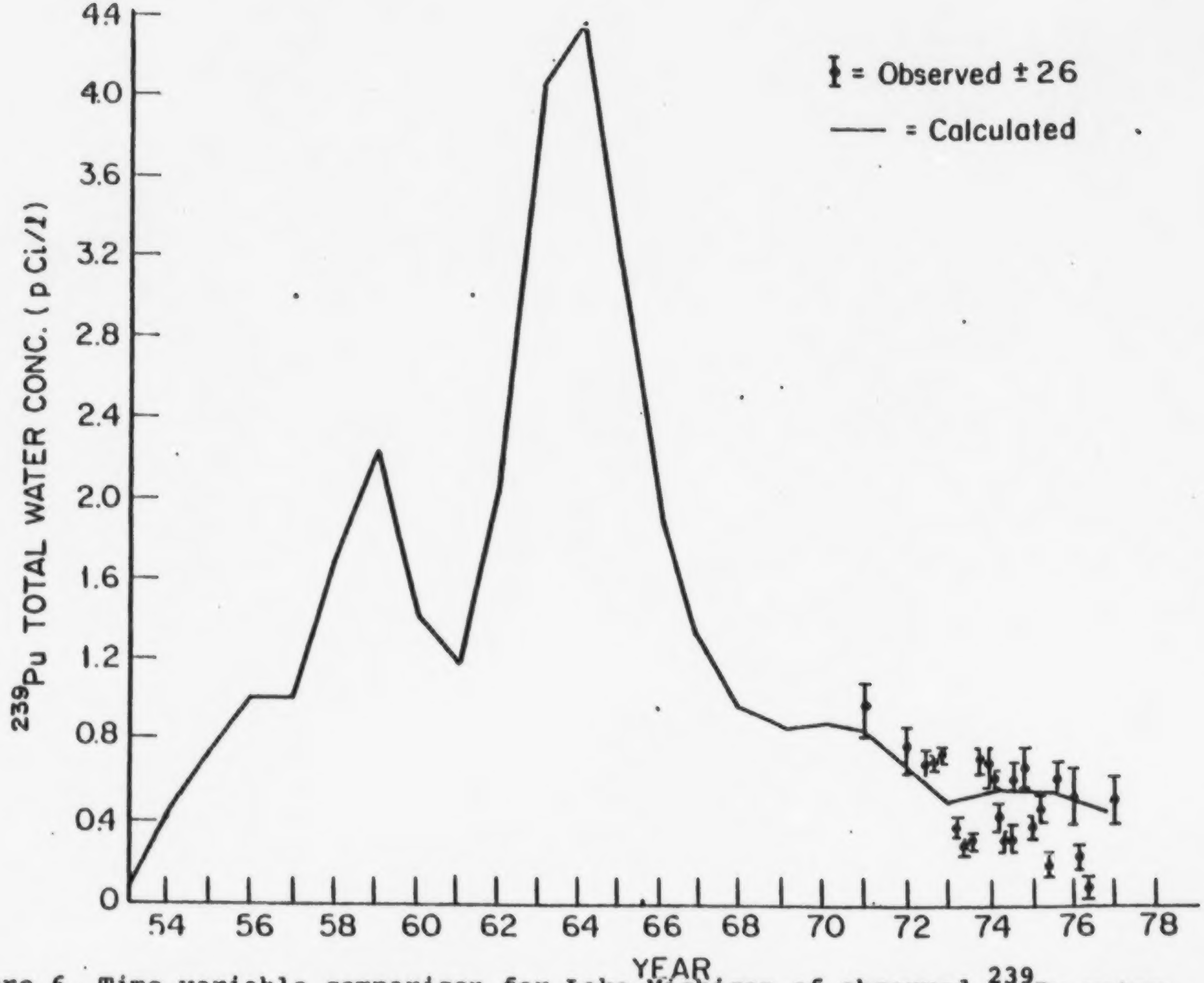


Figure 6. Time variable comparison for Lake Michigan of observed ^{239}Pu water column concentration and calculation for ^{239}Pu balance (From Thomann and DiToro, 1980). Data from Wahlgren et al. (1979).

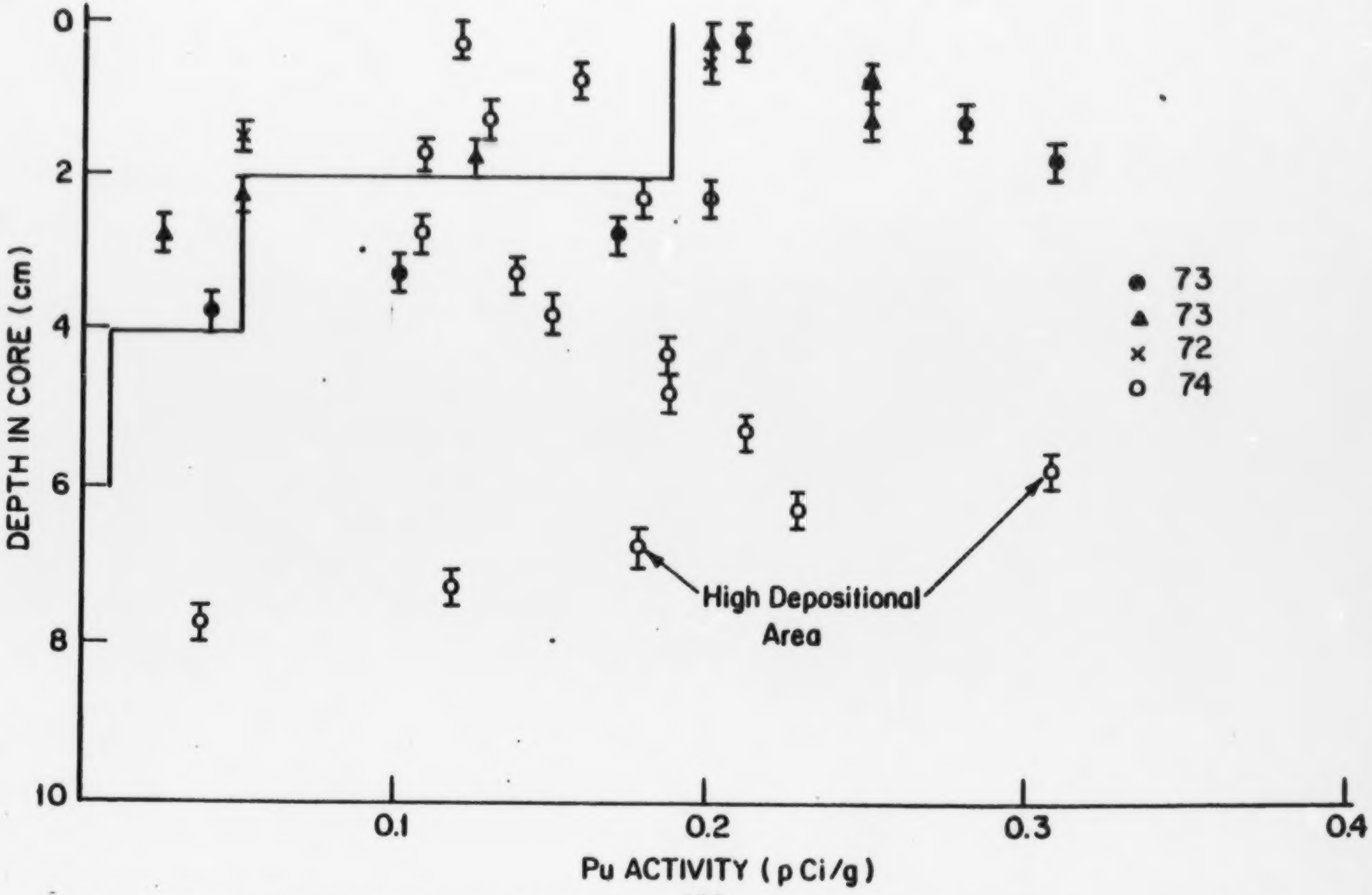


Figure 7. Comparison of observed sediment ^{239}Pu data (From Edington et al., 1976) and calculated values for 1973, segment 6, Lake Michigan (From Thomann and DiToro, 1980).

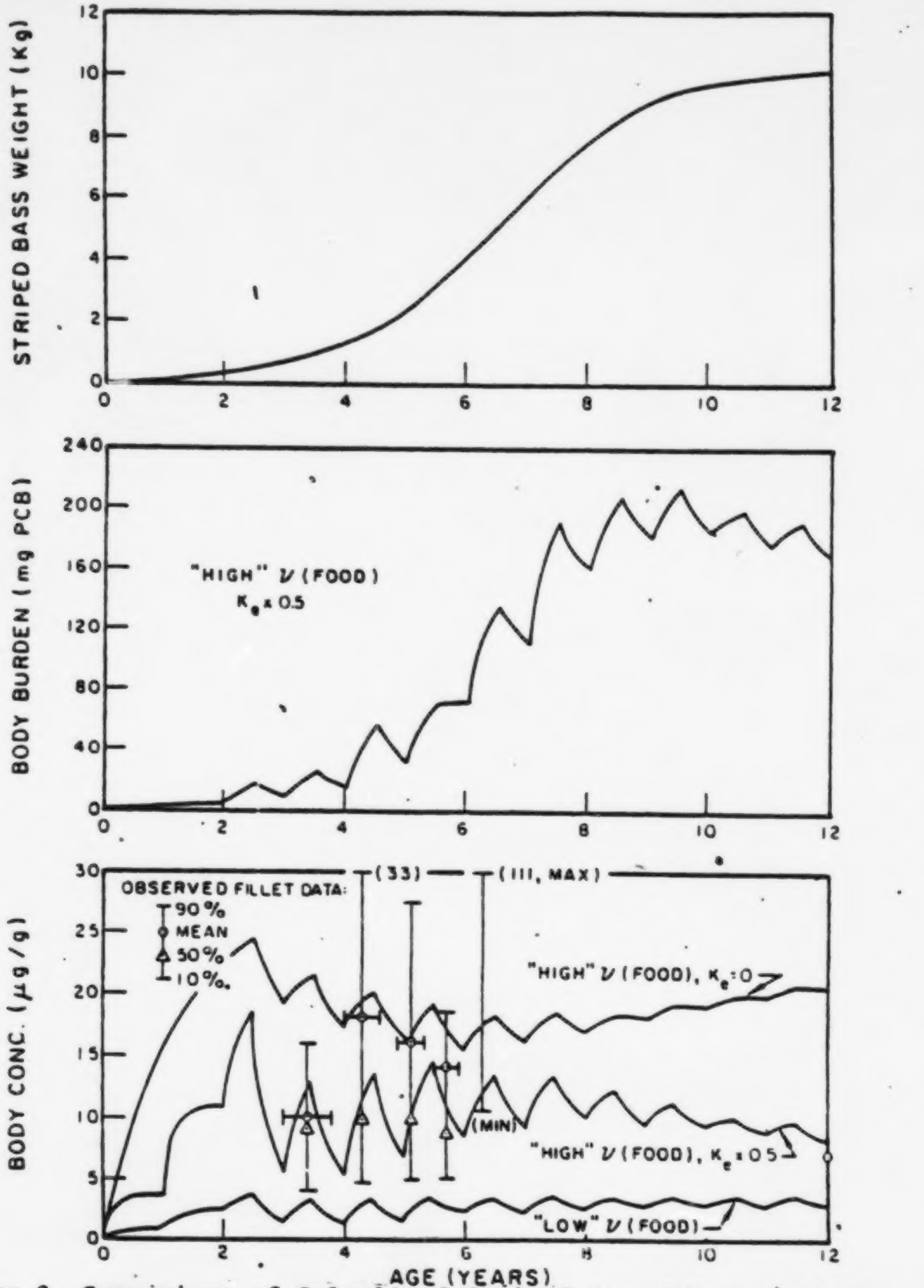


Figure 8. Comparison of Calculated Striped Bass PCB Concentrations Profiles with Observed Data (NYSDEC, Albany, N.Y.)

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APPENDIX

DEFINITION OF TERMS

Symbol	Definition	Units[1]
Q	Flow through water body	L^3_{s+w}/T
E'_{ij}	exchange rate of dissolved and/or particulate chemical	L^3_{s+w}/T
W	Mass input of dissolved chemical into water body	M/T
W_p	Mass input of particulate chemical into water body	M/T
W_T	Mass input of total chemical into water body	M/T
K_{11}	Sum of all first order loss rates in water (e.g. hydrolysis, evap., settling)	1/T
V_1	Volume of water plus solids	L^3_{w+s}
ϕ	Porosity = vol. water$_1$/(vol. water$_1$ + vol. solids$_1$)	L^3_w/L^3_{s+w}
M	solids concentration in water and/or sediment	M/L^3_{s+w}
π	partition coefficient (µg tox./g solids)/(µg tox./g water)	M/M÷M/M
v_{S1}	settling velocity of particles from water to sediment (active)	L/T
A	surface area of sediment/water interface	L^2
v_u	resuspension velocity of particles from sediment into water	L/T
v_s	settling or sedimentation rate of solids	L/T
f_D	fraction of total concentration in dissolved phase	-
f_p	fraction of total concentration in particulate phase	-

Symbol	Definition	Units[1]
c_D	dissolved concentration of chemical	M/L^3
c_P	particulate concentration of chemical	M/L^3_{s+w}
c_T	total concentration of chemical	M/L^3_{s+w}
λ	decay of dissolved chemical	1/T
k_u	sorption of chemical to particulate	$L^3/M\text{-}T$
K	desorption of chemical from particulate	1/T
L^3_{s+w}	volume of solids plus water	L^3
L^3_w	volume of water	L^3
ν	PCB concentration in striped bass	M/M
ν_i	PCB concentration in food of striped bass	M/M
k_{uf}	Uptake of dissolved PCB directly from water to fish	$L_w^3/M\text{-}T$
K_e	Excretion rate from fish	1/T
α	Assimilation efficiency of chemical absorption by fish from food	M/M
a'	Biomass conversion efficiency	M/M
C	Feeding rate of striped bass	M/M-T
w	Weight of striped bass	M
r	Weight loss rate; respiration, swimming	1/T

COMPARISON OF SIMPLE METHODS FOR PREDICTING THE FATE OF CHEMICALS IN THE CHESTER ESTUARY

Robert B. Ambrose and David W. Dallas

U.S. Environmental Protection Agency
College Station Road
Athens, GA 30605

ABSTRACT

Several simple methods for predicting chemical concentrations in the Chester Estuary are compared to a calibrated net variable flow simulation model. Manual formulas, steady-state models, and various modes of operating the net variable flow model are judged for their utility in screening promising treatment plant sites and in setting waste load allocations to protect given water quality standards. The formulas were found inadequate for calculating flushing times over the entire range flow of expected conditions. Steady-state methods and models were found adequate for screening analyses, but not reliable enough for setting waste load allocations. Net variable flow models, if properly calibrated, can be run in a steady state mode to set waste load allocations. Finally, conditions are defined for which simple mathematical treatment of chemical decay and partitioning is justified.

INTRODUCTION

Successful use of water quality criteria and standards depends on an ability to accurately translate pollutant loads into water quality concentrations. A wide range of techniques is available for this task, from simple formulas to complex simulation models. Many state pollution control agencies prefer using simple methods to set waste load allocations to meet their water quality standards, believing that the time and training required to operate more complex models is not cost- effective.

Many scientists and engineers, aware of the complexity of estuarine systems, however, warn that simple methods can lead to erroneous conclusions. It may actually take more training and experience to properly use and interpret results from simple methods than to use more complex but well-formulated simulation models.

In this paper, we describe how several simple methods compare in predicting chemical concentrations in the Chester Estuary. We take, as a case study, di-n-octylphthalate ester (DNOP). First, we compare three simple formulas for computing flushing with a more complex (though still simple) calibrated simulation model. Then, we compare two different modes of operating the simulation model: steady-state and variable, non-tidal flow. Finally, we compare the sensitivity of the variable flow simulation model to various environmental process parameters representing transport, degradation, and partitioning.

THE CHESTER RIVER ESTUARY

The Chester River on Maryland's Eastern Shore drains 1140 square kilometers of flat to gently rolling agricultural and wooded terrain typical of the Mid-Atlantic Coastal Plain. Following a meandering course, the tidal river broadens into a partially mixed, shallow estuary strongly influenced by tides, sediment, and water from the upper Chesapeake Bay (Clarke et al., 1972). A narrow main channel is bordered by broad submerged terraces that support commercially valuable shell fisheries. Sixty-four kilometers (km) below the head of tide, the 5-km-wide Chester River discharges an average volume of 12.7 cubic meters per second (m^3/sec) into the Chesapeake Bay.

The suspected source of DNOP in the Chester is a chemical factory in Chestertown, Maryland. This factory is permitted to discharge 2000 kilograms (kg) of total extractable organics per year into Morgan Creek, which empties into the Chester River at a point 53 km above the river's mouth. The waste chemicals first pass through a settling pond (residence time: 1000 days). Freeman and Cooney (1979) report an accumulation of 1.5×10^3 grams per cubic meter (g/m^3) of DNOP in the sediments of the pond, and a relatively constant 0.7 g/m^3 in the sediments throughout the Chester River. They conclude that although this chemical discharge is a possible source of the DNOP in the Chester River, it is not causally linked to past oyster mortality. They warn that the settling pond may soon reach saturation, however, resulting in serious accumulation and spreading of DNOP in the Morgan Creek area.

ANALYTICAL METHODS

The mixing and dispersal of chemicals throughout an estuary results from an often complex combination of advection and diffusion processes at different spatial and temporal scales. All methods of predicting chemical concentrations in estuaries are based on simplified representations of the mixing processes. Certain simplified mixing concepts are useful in providing the first indication of the degree of mixing expected in specific estuarine situations. Sometimes they may be sufficient in themselves to satisfy a particular engineering problem (Officer, 1976).

The flushing time of an estuary represents the length of time required to carry a pollutant out of an estuary under given steady flow conditions. Flushing time is typically defined as the time required to replace the existing fresh water in an estuary at a rate equal to the river discharge. Sometimes flushing time is taken as the time required to remove a pollutant introduced to the estuary at its head. These two definitions should be equivalent for estuaries with a single major inflow at the head. For estuaries like the Chester, with major inflows along its length, the flushing time computed under the two definitions can differ significantly.

We investigated several ways of calculating flushing times, including three simple formulas and a tidal-averaged simulation model operated in different modes. The tidal prism formula assumes the complete mixing of riverine and oceanic water each tidal cycle:

$$t_F = VT/P \qquad (1)$$

where t_F = flushing time, days
V = estuarine volume, m^3
T = tidal period, days
P = tidal prism (high water volume-low water volume), m^3

The modified tidal prism method (Ketchum, 1951) divides the estuary into segments of variable volume, such that

$$V_i = P_{i-1} + V_{i-1} \qquad (2)$$

where V_i = low water volume of segment "i", m^3

V_{i-1} = low water volume of segment upstream of "i", m^3

P_{i-1} = tidal prism of segment "i-1", m^3

The flushing time for each segment is then calculated by

$$T_i = (P_i + V_i) \times T/P_i \qquad (3)$$

The total estuarine flushing time is the sum of the times calculated for each segment. This method assumes the complete mixing of downstream tidal prism waters with the low water volumes of the upstream segment each tidal cycle. The fraction of fresh water method (Officer, 1976; Zison et al., 1977) uses observed salinity profiles to determine the fractional volume of fresh water residing in segments throughout estuary:

$$W_i = V_i(S_s - S_i)/S_s \qquad (4)$$

where W_i = volume of fresh water in segment "i", m^3

S_s = salinity concentration at seaward boundary, mg/m^3

S_i = salinity concentration in segment "i", mg/m^3

Flushing time for each segment can then be computed by

$$T_i = W_i \times T/R \qquad (5)$$

where R = freshwater inflow volume over tidal cycle, m^3

This method assumes that during each tidal cycle a volume of water equal to the river discharge moves into the downstream segment and mixes with the low water volume. The total estuarine flushing time is again the sum of the times calculated for each segment.

The simplest estuarine simulation model provides one-dimensional, steady-state solutions to a simplified mass balance equation:

$$\frac{\partial(UAC)}{\partial x} = \frac{\partial}{\partial x}\left(AE_{xs}\frac{\partial C}{\partial x}\right) \qquad (6)$$

where U = net, non-tidal velocity, m/sec

A = cross-sectional area, m^3

C = chemical concentration, mg/m^3

E_{xs} = steady-state longitudinal dispersion coefficient, m^2/sec

x = length, m

The dispersion coefficient E_{xs} incorporates the effects of all advective phenomena that contribute to mixing, such as tidal currents, density currents, lateral residual eddies, and embayment trapping.

Another simple estuarine simulation model provides variable flow, tidal-averaged solutions to the one-dimensional mass balance equation:

$$\frac{\partial(AC)}{\partial t} + \frac{\partial(UAC)}{\partial x} = \frac{\partial}{\partial x}\left(AE_{xt}\frac{\partial c}{\partial x}\right) \qquad (7)$$

where E_{xt} = tidal-averaged longitudinal dispersion coefficient, m^2/sec

t = time, sec

The dispersion coefficient E_{xt} is similar in concept and magnitude to E_{xs}.

In this analysis of simple estuarine transport models, we applied the Water Analysis Simulation Package, WASP (DiToro et al., 1978). WASP links mass transport computations based on equation 7 with a user-specified kinetic subroutine that is equivalent to a single compartment fate model.

Our 20-segment version of the WASP model (Figure 1) is based on a previously developed 85-segment network covering the tidal Chester River and Eastern Bay (Hatcher, 1975). We used the bathymetry developed by Hatcher when calibrating a steady-state version of WASP to salinity data. We estimated average monthly inflows to each segment between June 1975 and October 1977 by extrapolating the areal water yield from the USGS gage at Millington to the whole basin. Net advective flows in the estuary were solved by continuity. We calibrated dispersion coefficients using salinity data taken in the Chester between June 1975 and October 1977; the best fit was obtained with dispersion coefficients increasing from 72 m^2/sec at Millington to 480 m^2/sec at the confluence with Chesapeake Bay.

DISCUSSION OF RESULTS

COMPARISON OF FLUSHING TIMES

WASP was used to simulate a conservative substance in the Chester River at high and low flow in order to obtain a range of flushing times for the estuary. Steady loads were introduced at the boundaries of the major fresh water inputs. After steady state was reached, we applied a step input at the boundaries and ran the model until steady state was reached again. A graph of percentage response versus time is plotted in Figure 2. Because the system took a long time in reaching full steady state, we decided that when the mass reached 95

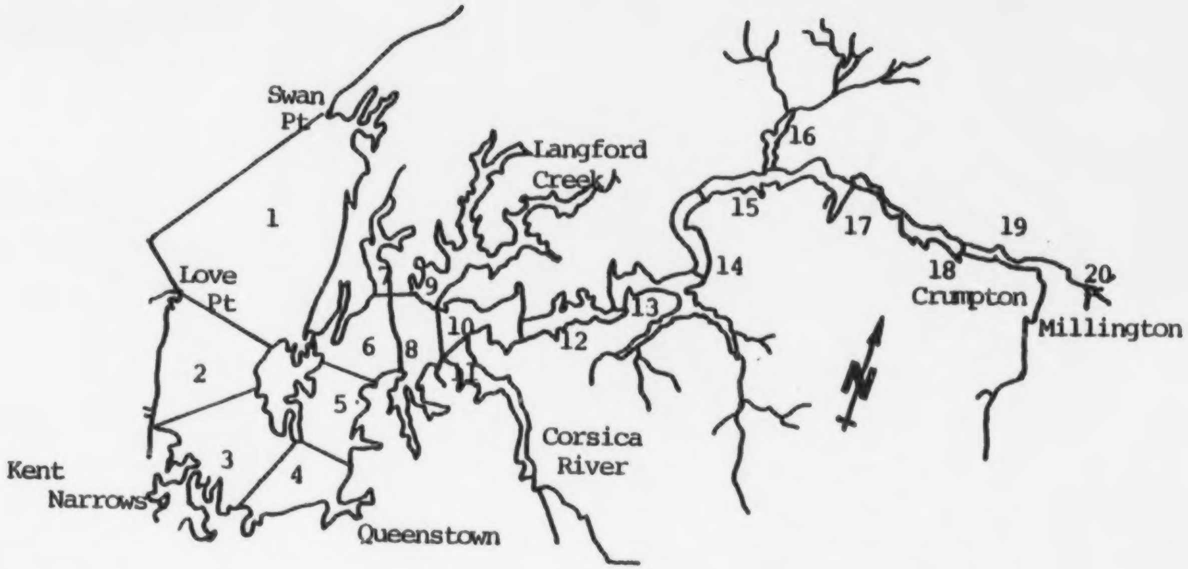
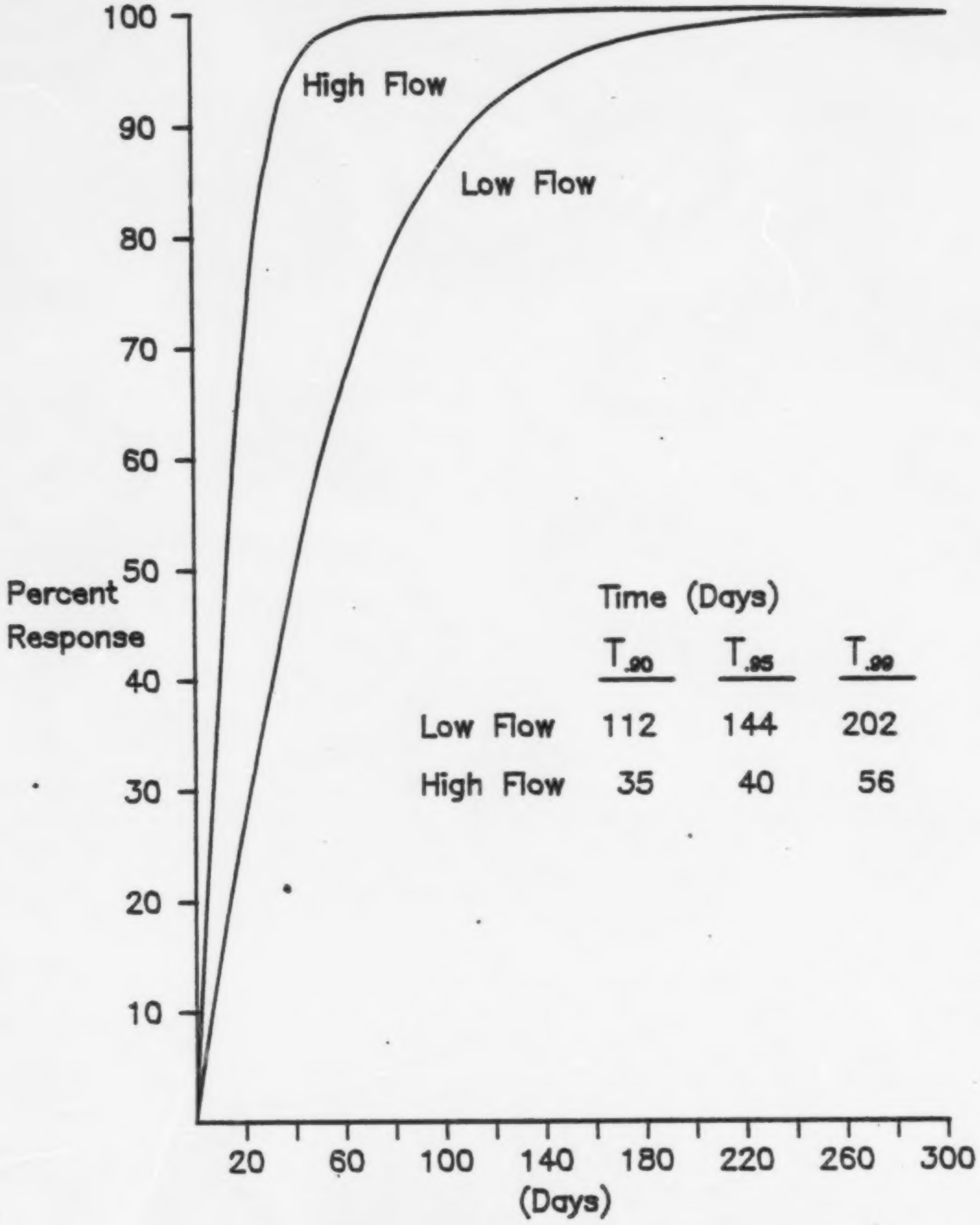


Figure 1. Chester River 20 segment network.



Time (Days)

	$T_{.90}$	$T_{.95}$	$T_{.99}$
Low Flow	112	144	202
High Flow	35	40	56

Figure 2. Percent response versus time for the Chester River.

percent of the steady state value, the system could be considered flushed.

We computed flushing times for five flows--0.12, 0.71, 1.41, 2.2 and 2.8 m^3/sec. Flushing time versus flow, plotted in Figure 3, is described by the following equation

$$y = 62.4\ x^{-0.40}$$

where y = flushing time, days
x = flow, m^3/sec
r = 1.0

We also used WASP to calculate the time required to remove a pollutant introduced at the head during low flow conditions. This alternate definition produced a flushing time of 171 days, a 20 percent increase.

Dean et al. (1980) computed the flushing time of the Chester using three simple manual techniques (Table 1). In comparing these formulas, we assume that the WASP conservative model calibrated to salinity data computes transport and flushing with good accurancy. The tidal prism method grossly underestimates flushing times and predicts no difference between high and low flow conditions. The basic assumption of this method--that ocean water completely mixes with the new river water each tidal cycle--is more appropriate for small tidal embayments than for an estuary as large as the Chester. The modified tidal prism method agrees very well with the WASP simulation for low flow, but differs significantly at high flow. The prediction that the flushing time is slightly longer for high flow than for low flow seems to be in error. The "fraction of fresh water" method agrees reasonably well with the WASP simulation for high flow, but overestimates low flow flushing by a factor of almost three.

FORMULA VERSUS MODEL CONCENTRATION PROFILES

These experiments compare the distribution of a conservative constituent of the WASP simulation with the fraction of freshwater (FFW) method used by Dean et al. (1980). The FFW, which distributes the chemical in the upstream direction as the salinity gradient and downestuary as the fraction of fresh water, was calculated for high flow conditions.

Five WASP simulations were run to steady state at the same high flow with a steady chemical load input to a segment of interest. Predicted concentrations from both WASP and FFM were normalized to the WASP input segment concentration.

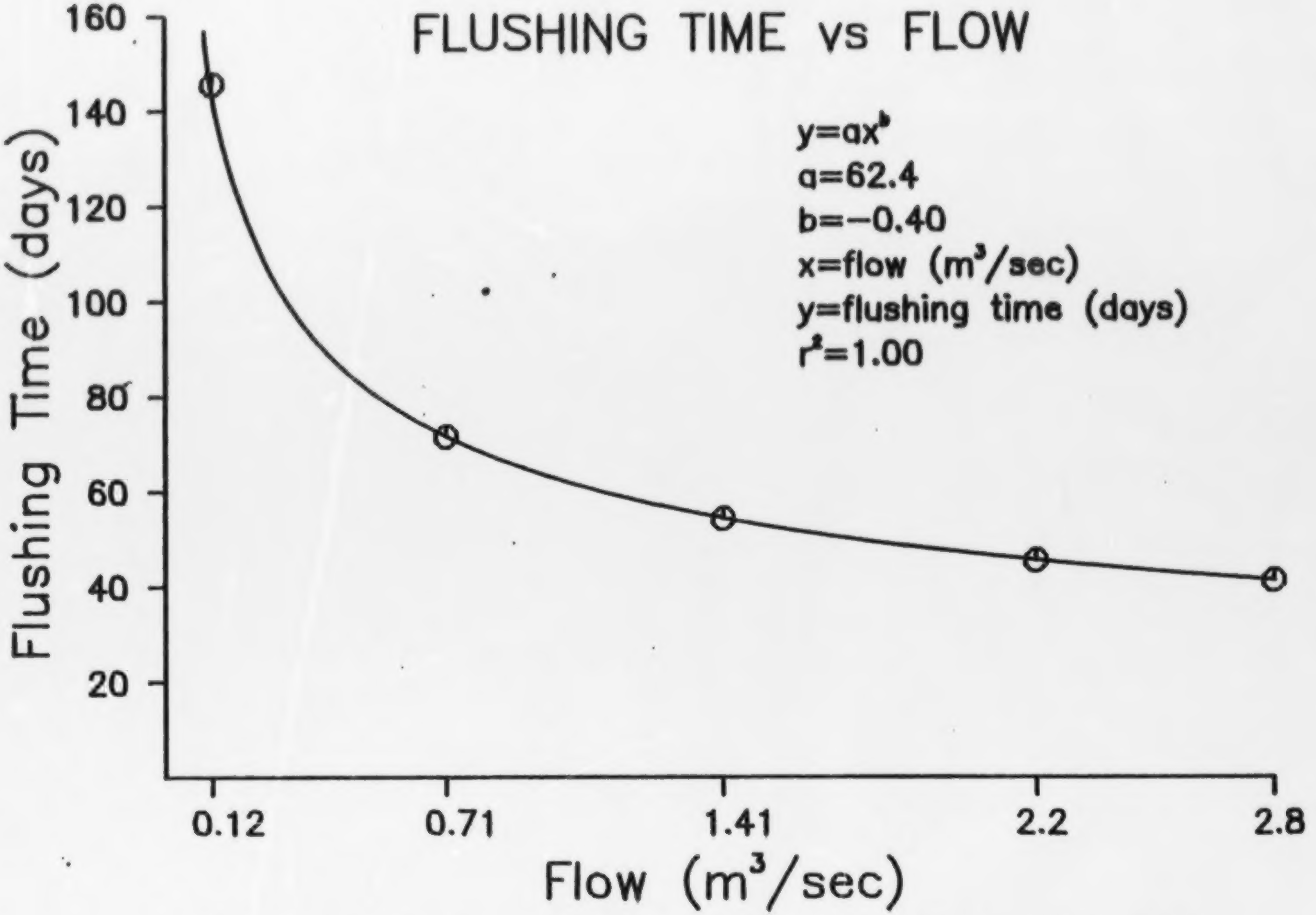


Figure 3. Flushing time versus flow for the Chester River.

Table 1

FLUSHING TIMES FOR THE CHESTER RIVER
COMPARISON OF FOUR METHODS

	Flushing Time (days)	
Method	High Flow	Low Flow
Tidal Prism	5.3	5.3
Modified Tidal Prism	143	134
Fraction of Freshwater	13.6	381
Net Flow Simulation	40	144

The WASP and FFM predictions for steady loads at the five segments can be compared in Figure 4. Each set of curves was analyzed using regression statistics and error calculations. The regression analyses were based on the standard equation $y = ax+b$, where x = WASP value and y = FFM value. (Results are shown in Table 2).

The FFM method incurred relative errors between 24 and 78 percent. It predicted lower maximum concentrations than WASP at the point of discharge, but higher concentrations upstream of the discharge segment. This would result when the observed salinity profile lies upstream of its high flow steady-state position. The FFW method distributes a chemical from the point of discharge downstream with the fraction of freshwater, here too low. The chemical is distributed upstream with the fraction of salt water, here too high. If the FFW method is used to predict steady-state low flow concentrations based on an observed salinity profile that was still intruding, these errors would be reversed. Predicted upstream concentrations should be too low, while concentrations at and below the point of discharge should be too high.

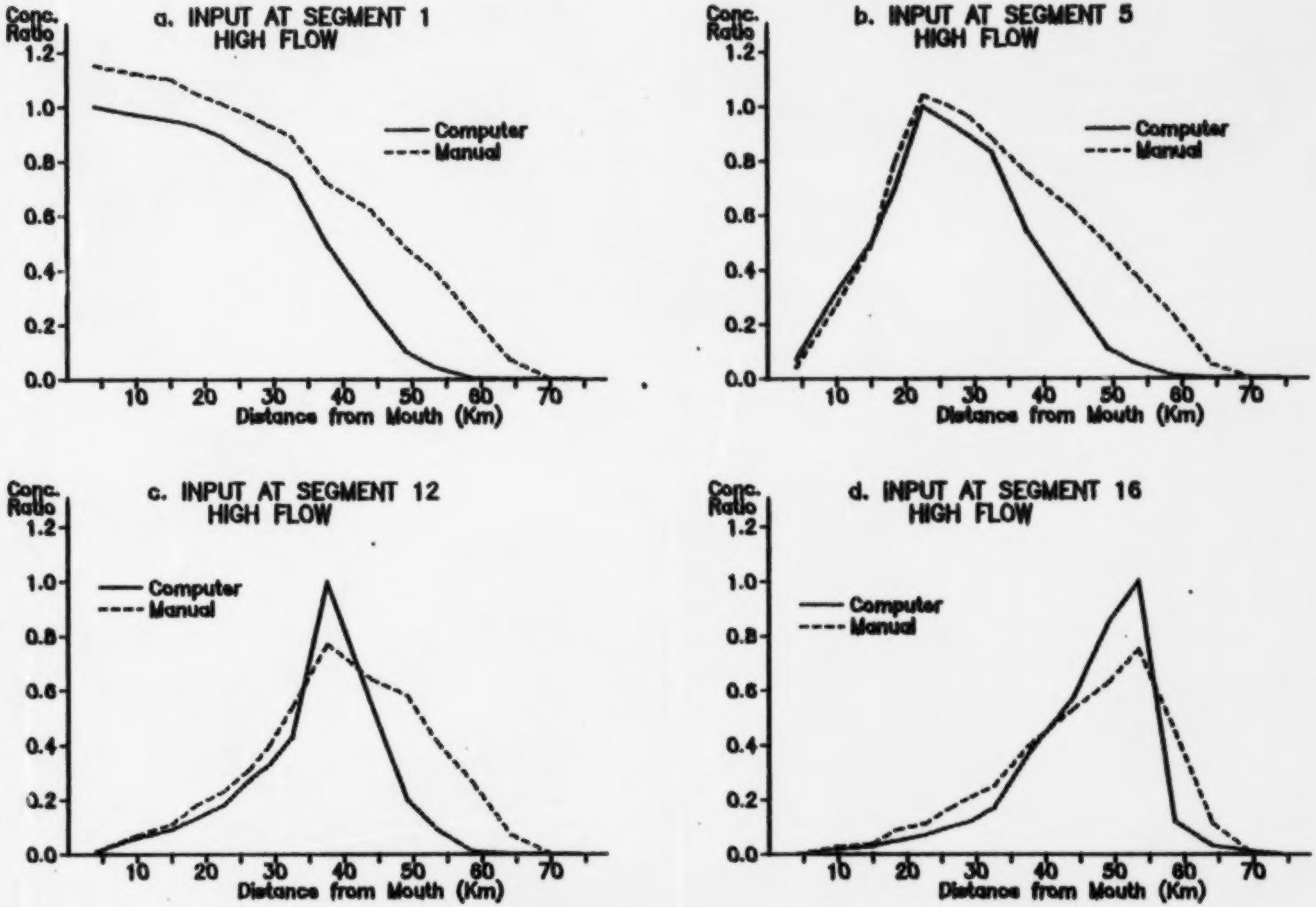


Figure 4. Concentration distribution—fraction of fresh water and steady state net flow for the Chester River.

Table 2

CONCENTRATION DISTRIBUTIONS - FRACTION OF FRESHWATER COMPARED TO STEADY NET FLOW SIMULATION

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.075	0.14	0.28	0.82	0.17	0.96
5	0.089	0.16	0.41	0.89	0.13	0.93
12	0.16	0.23	1.11	1.02	0.16	0.85
16	0.093	0.15	0.70	0.95	0.11	0.93
18	0.046	0.073	0.26	0.93	0.065	0.99

*Concentrations normalized to 1.0 at discharge segment

**High Flow conditions (2.8 cms at Millington)

STEADY-STATE VERSUS VARIABLE NON-TIDAL FLOW SIMULATIONS

Next, we compared variable non-tidal flow versus steady-state simulations. Perhaps the most important issue in using either is the choice of the tidal-averaged and steady state dispersion coefficients E_{xt} and E_{xs} (equations 7 and 6). Attempts have been made to derive predictable relationships between the salinity distribution and E_{xt} for individual estuaries. Harleman (1971) showed that where there is salinity-induced vertical circulation, the magnitudes of the derived disperison coefficients depend on arbitrary assumptions and are not predictable. Attempts to correlate dispersion to distance and freshwater inflow also have pitfalls, particularly in cases where the estuary may take a period of months to adjust to short term changes in freshwater discharge (Ward and Fischer, 1971). The problems in estimating proper values for E_{xt} can be controlled, however, if the model is calibrated and tested with data of sufficient duration and spatial coverage. Whether predictable correlations for E_{xt} could be found, at least the magnitude of dispersive transport could at least be bracketed for the period of record.

Because an extensive salinity data set was not availabe in the Chester, we derived only one set of spatially variable dispersion coefficients through calibration. The varying freshwater inflow and salinity boundary conditions are shown in Figure 5. We calibrated the salinity profile computed for September 15, 1976, to data taken along the Chester on that date. The best fit, shown in Figure 6a, was obtained with values of E_{xt} increasing from 72 m^2/sec at Millington (section 20) to 480 m^2/sec at the mouth (section 1).

The derivation of disperison coefficients E_{xs} for steady-state simulations is complicated by the rarity of actual steady-state conditions in estuaries. The flushing time of the Chester for the 0.3 m^3/sec flow recorded at Millington during September 1976 is estimated to be 100 days. To determine how much error would be introduced by assuming steady-state conditions, we recalibrated WASP to the September 15 data in a steady-state mode.

The fit shown in Figure 6b was calibrated with E_{xs} values to 50 percent of the E_{xt} values. A steady-state simulation using the more nearly correct E_{xt} values gives the profile shown in Figure 6c. This profile is an estimate of how far observed September 1976 salinity concentrations differed from the steady-state concentrations expected after 3 to 4 months of this low flow. A statistical comparison of the incorrect steady-state profile (6b) with the correct steady-state profile (6c) gives standard and relative errors of 0.73 kg/m^3 and 11 percent. Linear regression gives $Y = 1.12x - 1.34$, with a correlation coefficient of 0.99. The observed salinity profile was apparently not far from its steady-state position for 0.3 m^3/sec, probably because of the long preceding period of gradually declining flows.

Next we compared variable non-tidal flow versus steady-state simulations using both the correctly and the incorrectly calibrated dispersion coefficients. We wanted to find (1) how steady-state simulations at the high, low, and median flows approximate the low, high, and median concentration profiles from a time-variable simulation, and (2) how much error is introduced by using the improperly calibrated dispersion coefficients. Our time variable simulations covered the period June 1975 through October 1977 with a 0.1 day time step, zero initial conditions, zero boundary conditions, and average monthly flows. A steady chemical load was applied to the segment under investigation. Minimum simulated concentrations generally occurred during a high flow period in January 1976. Maximum concentrations occurred during October 1977 which followed a prolonged dry period. The model was then run at the high, low, and median flows from the 845 day period until steady state was reached.

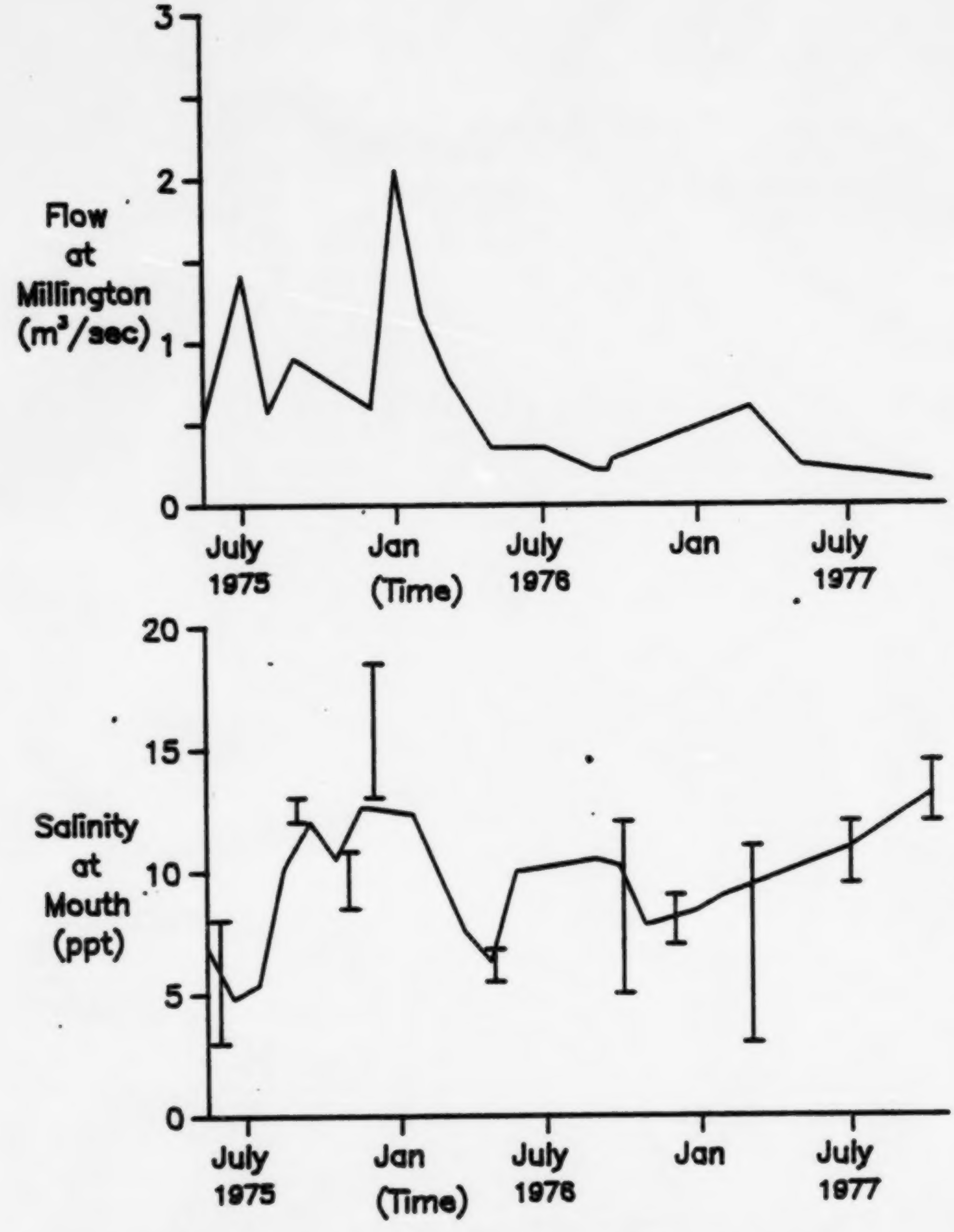
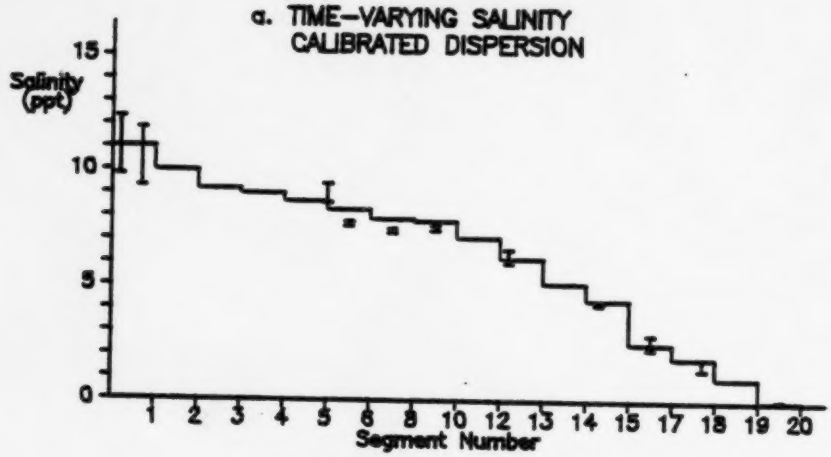
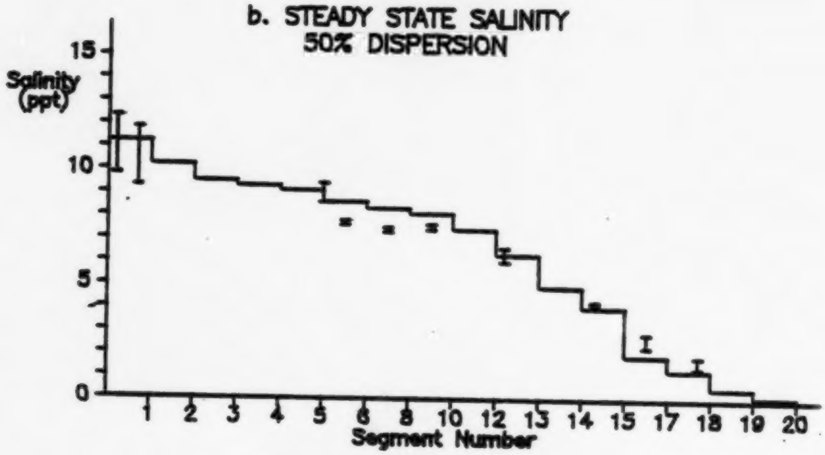


Figure 5. Calibration conditions for time–varying net flow model.


a. TIME-VARYING SALINITY CALIBRATED DISPERSION


b. STEADY STATE SALINITY 50% DISPERSION

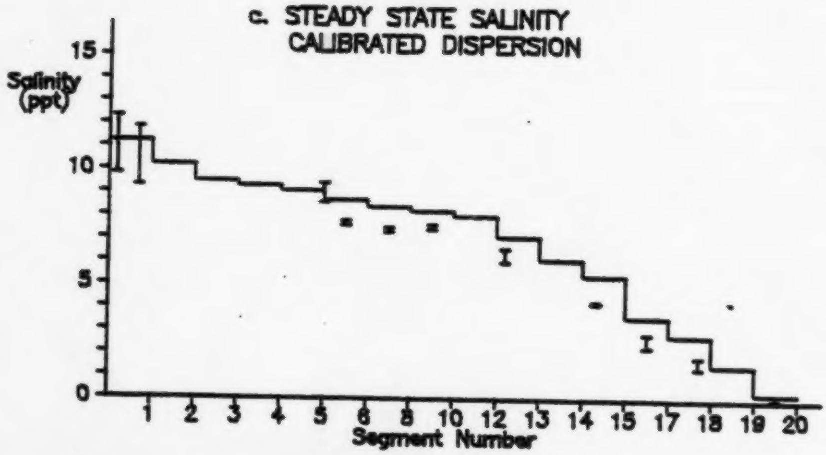

c. STEADY STATE SALINITY CALIBRATED DISPERSION

Figure 6. Steady state and time-varying calibration results for the Chester River.

The "correct dispersion" steady concentration profiles and variable concentration ranges are compared in Figure 7.

Statistical analyses for the "correct dispersion" and the "incorrect dispersion" steady state concentrations, calculated against the time variable simulation results, are given in Tables 3 and 4. The importance of attaining accurate dispersion coefficients is evident. Whereas the relative errors for the "correct dispersion" steady simulations range between 8 percent and 31 percent, the "incorrect dispersion" steady simulations incur relative errors between 27 percent and 110 percent. The average errors show that steady-state predictions are generally high--from 0 to 0.1 for the "correct dispersion" simulations, and from 0 to 0.6 for the "incorrect dispersion" simulations. The steady-state simulations become more accurate as the discharge location is moved upstream. This could result from the more rapid flushing of the upper reaches, allowing the changing concentration distribution to more closely approach steady-state conditions.

Table 3

CONCENTRATION DISTRIBUTIONS-STEADY-STATE,

INCORRECT DISPERSION TO VARIABLE NET FLOW SIMULATIONS

A. Low Flow - High Concentration

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.63	0.73	1.0	2.3	-0.29	0.99
5	0.51	0.62	1.0	2.1	-0.15	0.98
12	0.27	0.41	1.1	1.8	-0.040	0.98
16	0.15	0.28	1.1	1.7	-0.019	0.98
18	0.11	0.20	0.98	1.5	0.010	0.98

Table 3 (cont.)

B. Median Flow - Median Concentration

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.50	0.65	1.0	2.2	-0.26	0.98
5	0.38	0.54	1.0	2.0	-0.15	0.97
12	0.14	0.27	0.93	1.6	-0.032	0.96
16	0.056	0.14	0.81	1.4	-0.010	0.97
18	0.027	0.053	0.44	1.1	0.016	0.98

C. High Flow - Low Concentration

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.34	0.53	1.0	1.9	-0.14	0.97
5	0.25	0.40	1.0	1.8	-0.078	0.97
12	0.06	0.12	0.77	1.3	0.012	0.94
16	0.0073	0.033	0.43	0.96	0.010	0.96
18	0.0046	0.019	0.27	0.78	0.020	0.97

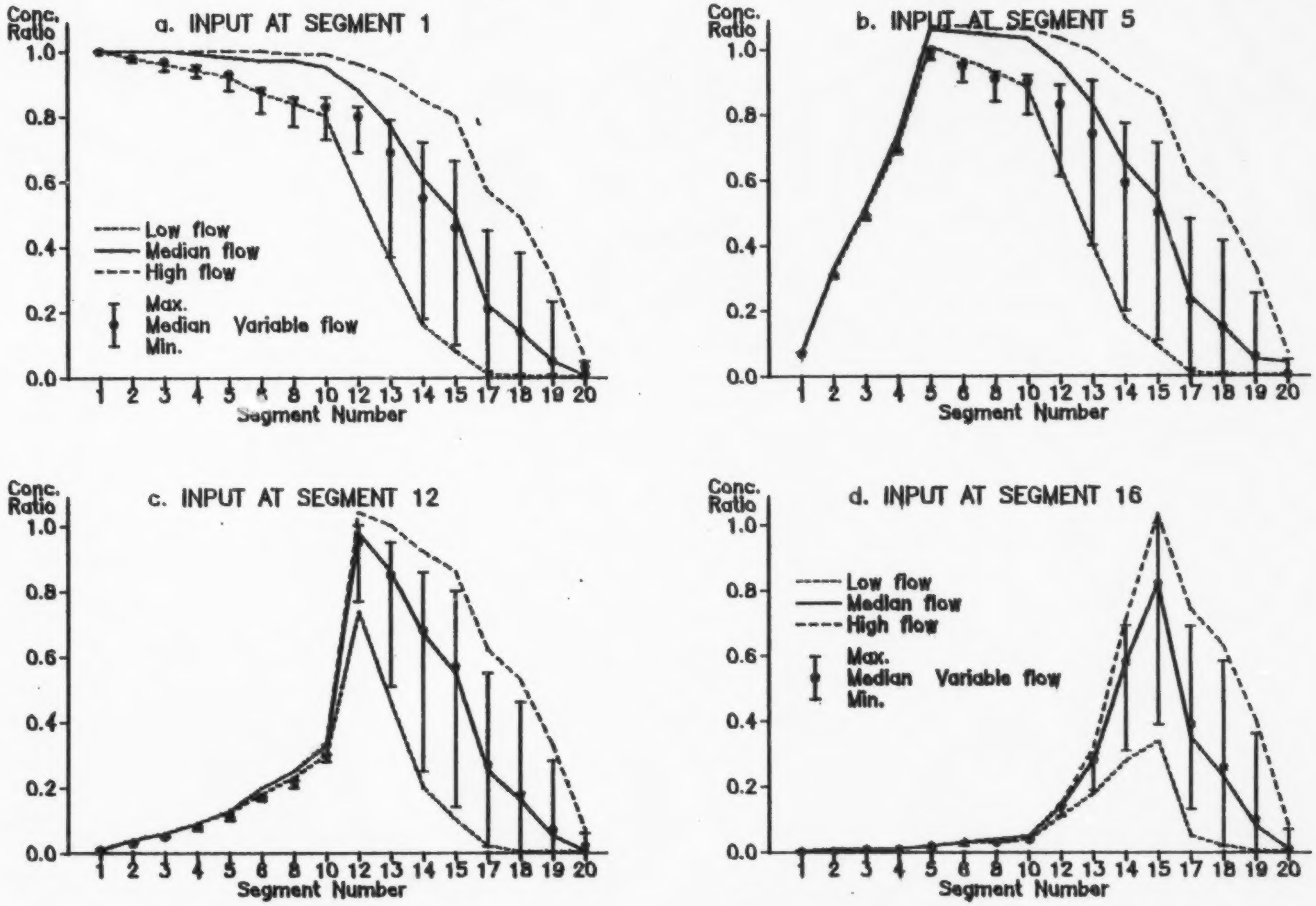


Figure 7. Concentration profiles for the Chester River.

Table 4

CONCENTRATION DISTRIBUTIONS-STEADY-STATE, CORRECT DISPERSION TO VARIABLE NET FLOW SIMULATIONS

A. Low Flow - High Concentration

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.086	0.099	0.14	0.99	0.093	0.98
5	0.088	0.10	0.16	1.10	0.024	1.0
12	0.0069	0.12	0.30	1.02	-0.0011	0.95
16	0.015	0.023	0.092	1.05	0.0030	1.0
18	0.013	0.021	0.10	1.05	0.0021	1.0

B. Median Flow - Median Concentration

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.045	0.062	0.096	1.07	0.0022	0.99
5	0.052	0.071	0.13	1.13	-0.019	1.0
12	0.0015	0.022	0.078	1.02	-0.0029	1.0
16	-0.0046	0.016	0.091	0.97	-0.00084	1.0
18	0.0049	0.015	0.13	0.93	0.0033	1.0

Table 4 (cont.)

C. High Flow - Low Concentration

Discharge Segment	Errors			Regression Statistics		
	Average	Standard	Relative	Slope	Intercept	Correlation
1	0.012	0.046	0.088	1.02	0.0023	0.99
5	0.020	0.035	0.10	1.09	-0.014	1.0
12	-0.0022	0.025	0.16	0.94	0.0079	0.99
16	-0.0094	0.025	0.31	0.87	0.0019	0.99
18	-0.0065	0.014	0.30	0.83	0.0014	0.99

SENSITIVITY TO BASIC ENVIRONMENTAL PROCESSES

In the last part of this study, we investigate the sensitivity of the variable flow simulation model to various environmental processes representing transport, degradation, and partitioning. Our basic case study, reported in Bostater et al. (1980), simulates the distribution of 2000 kg/year of DNOP from Morgan Creek (segment 16) during the period June 1975 to October 1977. The model assumes that bottom sediments are in equilibrium with suspended sediments; that particulate adsorbed and biologically adsorbed DNOP is in continuous equilibrium with dissolved DNOP; and that DNOP is lost only through advective and dispersive transport, alkaline hydrolysis, and microbial decay. Transport equation 7 is solved for total DNOP with the following sink term added to the right hand side:

$$S = -(KB + KH)\,C_W \qquad (8)$$

where C_W = dissolved DNOP, mg/m^3
K_B = bacterial degradation, day^{-1}
= KM x BPOP
K_H = alkaline hydrolysis decay, day^{-1}
= $KOH \times 10^{(PH-14)}$

Other definitions, units and values, reported in Wolfe et al., 1979, are given in Table 5.

We partitioned the total DNOP (C_T, mg/m^3) into dissolved, particulate (C_p, mg/m^3), and biological (C_B, mg/m^3) phases with the following equations.

$$C_w = \frac{C_T}{1 + KP \times SS \times 10^{-6} + KO \times BM \times 10^{-6}} \qquad (9)$$

$$C_p = \frac{C_T \times SS \times KP \times 10^{-6}}{1 + KP \times SS \times 10^{-6} + KO \times BM \times 10^{-6}} \qquad (10)$$

$$C_B = \frac{C_T \times BM \times KO \times 10^{-6}}{1 + KP \times SS \times 10^{-6} + KP \times BM \times 10^{-6}} \qquad (11)$$

Concentrations per unit sediment (G_P, mg/kg) and per unit biomass (G_B, mg/kg) can be calculated from the volumetric concentrations as follows.

$$G_p = C_p/(SS \times 10^{-6}) \qquad (12)$$

$$G_B = C_B/(BM \times 10^{-6}) \qquad (13)$$

Predicted total DNOP concentrations, reported in Bostater et al. (1980), follow the distribution in Figure 7d. The sensitivity analysis, presented in Table 6, demonstrates the major influence of dispersive transport on total DNOP concentrations throughout much of the Chester. Boundary exchange was most important near the mouth, whereas net advective flow was most important near the head. Large changes in the net decay rate of 5 x 10^{-4} day^{-1} alter DNOP concentrations only slightly. Partitioning has little effect on total DNOP, because low sediment concentrations drive most of the mass into the dissolved phase.

Table 6 shows that for decay processes to affect concentrations as much as transport processes, the effective first order rate must exceed 0.01 day^{-1}. The matrix in Table 7 shows how the second order chemical decay rates and estuarine environmental characteristics interact to produce effective first order microbial and alkaline hydrolysis decay rates. For DNOP, with a microbial degradation rate of 7.4 x 10^{-12} l org^{-1} day^{-1}, the effective bacterial population must exceed 10^7 organisms/ml before microbial decay becomes very important. The alkaline hydrolysis rate of 27 Mole^{-1} day^{-1} is low enough to prevent alkaline hydrolysis from becoming important even at pH 10. At the estimated bacterial population and pH of the Chester, a chemical would have to

Table 5
MODEL DNOP PARAMETERS

Constant	Description	Units	Value
KOW	Octonal-water partition coefficient		1.6×10^4
OCC	Lipid fraction of oysters	g fat/g edible	.02
KO	Effective oyster-water partition coefficient		320
BM	Benthic oyster biomass	g edible/m^2	1.0
KPO	Organic sediment-water partition coefficient		5.9×10^4
OC	Organic fraction of sediment (wet basis)	g org/g sed	0.05
KP	Effective sediment-water partition coefficient		590
SS	Suspended sediment	g/m^3	30
KOH	Alkaline hydrolysis rate	1/Molar/day	27
PH	Hydrogen Ion Activity	Std Units	7.7
KH	Effective hydrolysis rate	1/day	1×10^{-5}
KM	Microbial degradation rate	1/org/day	7.4×10^{-12}
BP	Sediment bacterial population	org/ml	1.0×10^6
BPOP	Effective depth average bacterial population	org/ml	10^4-10^5
KB	Effective bacterial degradation rate	1/day	2-5×10^{-4}
KD	Effective decay rate	1/day	2-5×10^{-4}

Table 6

PREDICTED SENSITIVITY TO 50 PERCENT REDUCTION IN PROCESS MODEL PARAMETERS

Model Parameters Reduced 50 Percent	Percent Concentration Change to Segment 2	5	12	15	18
Transport Processes[1]					
dispersion	+ 90	+ 90	+ 85	+ 70	+ 8
boundary exchange	+ 21	+ 6	+ 1	0	0
net advection	+ 1	+ 1	+ 2	+ 8	+ 37
Decay Processes[1] (net first order decay, day^{-1})					
from 0.001	+ 2	+ 2	+ 3	+ 1	+ 3
from 0.010	+ 24	+ 23	+ 19	+ 12	+ 24
from 0.100	+ 252	+ 221	+141	+ 59	+169
from 1.000	+5517	+2533	+929	+181	+112
Partitioning Processes					
suspended sediment[2]	+ 4	+ 4	+ 4	+ 4	+ 4
organic fraction[2]	+ 4	+ 4	+ 4	+ 4	+ 4
partition coefficient[2]	+ 4	+ 4	+ 4	+ 4	+ 4
suspended sediment[3]	+ 4	+ 4	+ 4	+ 4	+ 4
organic fraction[3]	- 48	- 48	- 48	- 48	- 48
partition coefficient[3]	- 48	- 48	- 48	- 48	- 48

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[1] Total concentration

[2] Dissolved fraction

[3] Sediment fraction

have microbial and alkaline hydrolysis decay rates of 10^{-10} l org^{-1} day^{-1} and 10^4 Mole^{-1} day^{-1} before the decay process compete with the transport processes for importance.

The accuracy of this simulation depends, in part, on how closely DNOP follows the same transport processes as salinity, which was used for calibration. When most of the chemical is in the dissolved phase, the simulation should be relatively accurate. When a large fraction of the chemical is sorbed, however, sediment transport processes not included in this simulation can significantly affect total concentrations. The matrix in Table 8, shows how the organic sediment-water partition coefficient and the suspended sediment concentration affect the fraction of the chemical in the dissolved phase. For DNOP, with a partition coefficient of 59,000, more than 90 percent is dissolved at an average suspended sediment concentration of 30 g/m^3. If sediment concentrations quadrupled to about 120 g/m^3 (with the sediment organic content remaining a constant 5 percent), then 25 percent of the chemical would be sorbed, and sediment transport could be considered significant. On the other hand, if the organic fraction fell to 1 percent, more than 90 percent of DNOP would remain dissolved at sediment concentrations as high as 150 g/m^3.

Table 7

DECAY SENSITIVITY

a. Microbial Decay

Microbial Degradation Rate	Bacterial Population, organisms/ml			
	10^5	10^6	10^7	10^8
1/org/day				
10^{-13}	10^{-6}*	10^{-5}	10^{-4}	10^{-3}
10^{-12}	10^{-5}	10^{-4}	10^{-3}	10^{-2}
10^{-11}	10^{-4}	10^{-3}	10^{-2}	10^{-1}
10^{-10}	10^{-3}	10^{-2}	10^{-1}	10^{0}

b. Alkaline Hydrolysis Decay

Alkaline Hydrolysis Degradation Rate	Hydrogen Ion Activity (pH)			
	7	8	9	10
1/M/day				
10^{1}	10^{-6}*	10^{-5}	10^{-4}	10^{-3}
10^{2}	10^{-5}	10^{-4}	10^{-3}	10^{-2}
10^{3}	10^{-4}	10^{-3}	10^{-2}	10^{-1}
10^{4}	10^{-3}	10^{-2}	10^{-1}	10^{0}

*Effective first order decay rate, day^{-1}

Table 8
PARTITIONING SENSITIVITY

Organic sediment-water partition coefficient (KPO)	Suspended Sediment Concentrations, g/m^3					
	10	50	100	500	1000	5,000
200	.9999	.9995	.9990	.9950	.9901	.9524
1000	.9995	.9975*	.9950	.9756	.9524	.8000
2000	.9990	.9950	.9901	.9524	.9091	.6667
10,000	.9950	.9756	.9524	.8000	.6667	.2857
20,000	.9901	.9524	.9091	.6667	.5000	.1667
100,000	.9524	.8000	.6667	.2857	.1667	.0385
200,000	.9091	.6667	.5000	.1667	.0909	.0196
1,000,000	.6667	.2857	.1167	.0385	.0196	.0040

*Fraction of chemical in dissolved phase; assumes sediment organic content of 5 percent (KP = 0.05 KPO)

This simple chemical simulation assumes that scouring of bed sediments is minimal throughout the Chester and that exchange of chemicals between the bed and the overlying water is slow relative to the transport of overlying waters. Although the bed would store a large amount of chemical, it should not affect equilibrium concentrations in the overlying water. This simulation, then, should be relatively accurate in predicting the estuarine response to a long-term steady chemical load. Abrupt changes in either the sediment or the loading regimes could not be simulated without accounting for exchanges with the bed.

SUMMARY AND CONCLUSIONS

We set out to compare several simple methods for predicting chemical concentrations in the Chester Estuary. We looked at manual formulas, steady-state models, and various modes of operating net variable flow simulation models. The latter was calibrated to one set of salinity data and used as our standard for comparison. Although this standard is itself too simple to allow absolute judgment about the accuracy of the simpler methods, we can determine how reliably they could duplicate its two principal functions: screening estuaries for promising treatment plant sites and setting waste load allocations to protect given water quality standards.

Our main conclusions, summarized in more detail below, are as follows. None of the manual methods for calculating flushing times could be used to reliably screen the Chester over its entire range of expected flow conditions. Steady-state methods and simulation models, which are based on observed data presumed to be at steady state, are useful for quick screening analyses but are not reliable enough for setting waste load allocations. Net variable flow models, if properly calibrated in time-varying simulations, can be run in a steady mode to set waste load allocations. Simple mathematical treatment of chemical decay and partitioning is justified for finding the equilibrium concentrations of certain classes of chemicals. Other classes of chemicals require more sophisticated handling of decay or sediment transport processes. Calculating the response to abrupt changes in chemical loading or sediment regimes requires a model that accounts for exchanges with the bed sediments.

Flushing time is an important characteristic to consider when screening estuaries to locate potential treatment plant sites. Flushing time represents the length of time required to carry a pollutant out of an estuary under given steady flow conditions. Of the tidal prism method, the modified tidal prism method, and the fraction of fresh water method, only the modified tidal prism method works well for low flow conditions. None of the methods could be considered reliable when applied to an estuary such as the Chester over a range of conditions.

Even if steady-state conditions never fully develop in many estuaries, theoretical steady-state predictions are often considered appropriate for screening potential treatment plant sites, and even for determining waste load allocations. One of the simplest steady-state methods is the manual fraction of fresh water method.

Maximum concentrations computed by fraction of fresh water for discharges away from the seaward boundary fell below those produced by net-flow simulations run to steady-state. Concentrations above the point of discharge, however, were overpredicted, and relative errors ranged between 24 and 78 percent. For low flow conditions, the fraction of fresh water method is expected to predict maximum concentrations that are too high, and upstream concentrations that are too low. Actual errors will depend on how far the observed salinity profile differs from its steady-state location at the given flow. We conclude that the fraction of fresh water method can be used as a screening tool for comparing various discharge locations under the same flow and salinity conditions. Extreme caution should be used when comparing different estuaries, or even different flow conditions within the same estuary. Using this method to set waste load allocations could result in an unknown overdesign or underdesign of treatment facilities to meet a given standard.

A major problem with using steady-state models in estuaries is quantifying dispersive transport. Dispersion coefficients must be calibrated with salinity data that seldom represent true steady-state conditions in estuaries like the Chester. These incorrectly calibrated dispersion coefficients, then, introduce an unknown error, the magnitude of which depends on how far the observed salinity profile lies from its theoretical steady-state position. In this case study, the data were gathered under conditions reasonably close to steady-state. When calculated concentrations for discharges under low, median, and high flow conditions were compared with high, median, and low concentrations from varying flow simulations, relative errors between 27 percent and 110 percent were obtained. We conclude that, although steady-state models could be used as screening tools, they are not reliable enough for setting waste load allocations to meet given water quality standards in estuaries such as the Chester.

If dispersive transport is properly calibrated with a time- varying model, however, steady simulations do a good job bounding the concentrations predicted from a varying flow simulation. Given the correctly calibrated dispersion coefficients, relative errors for the steady-state concentrations were reduced to between 8 percent and 31 percent. Particularly good results were obtained for the upper reaches of the Chester, where short flushing times made the steady-state approximation more valid. In the lower reaches, the flushing times are much longer; the steady-state approximation consistently overestimates concentrations due to chemical discharges into the lower reaches. We conclude that steady simulations with appropriate dispersion coefficients are acceptable for setting waste load allocations. We note that properly calibrating the dispersion coefficients requires a time variable model.

When simulating a chemical such as DNOP, the transport process of dispersion is most important, followed by boundary exchange near the mouth and net advective flow near the head. Microbial degradation and alkaline hydrolysis should not be important processes in the Chester for chemicals with decay rates less than 10^{-10} l organism^{-1} day^{-1} and 10^4 Mole^{-1} day^{-1}. Sediment partitioning and transport could become important for DNOP in suspended sediment regimes exceeding 120 g/m^3. At average 30 g/m^3 suspended sediment concentrations with a 5 percent organic content, partition coefficients much higher than 2 x 10^5 should cause sediment transport processes to significantly affect the total chemical concentration profile. In that case, a more sophisticated model is needed to set waste load allocations.

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BOTTOM SEDIMENTS AS A SOURCE OF NUTRIENTS AND TRACE ELEMENTS IN THE ESTUARINE ENVIRONMENT

Owen P. Bricker

EPA/Chesapeake Bay Program
2083 West Street
Annapolis, MD 21401

Bottom Sediments as a Source of Nutrients and Trace Elements in the Estuarine Environment

INTRODUCTION

Estuaries receive inputs of dissolved and particulate matter from tributary rivers, shoreline erosion, the coastal marine environment, the atmosphere and the biosphere. This matter consists of the natural products of weathering and erosional processes and of biologic activity. In addition, ever increasing amounts of byproducts and wastes produced by the activities of man enter the estuary either directly in discharge and effluent streams or via nonpoint source runoff (Fig. 1). A large proportion of both the natural and the anthropogenic matter is strongly associated with sediments, particularly those of fine particle size and high surface area. These sediments are transported to various parts of the estuary where they settle and accumulate on the bottom. The sediments blanketing the bottom of the estuary constitute the largest reservoir of chemical substances in the estuarine environment. The behavior of sediment-associated constituents after burial will determine their fate in the system. If they remain with the sediment they may be, for all practical purposes, removed from the system. If, however, they are mobilized by processes active within the sediment they may become available to the benthic fauna and may also be returned to the water column. This has implications particularly for toxic materials and nutrients. To assess the importance of bottom sediments as a source of these substances it is necessary to understand the chemical, physical and biological processes that occur in the benthic sediment environment.

Benthic sediment environment

Within the bottom sediments the chemical and physical conditions are very different from those in the water above. Sediment particles settle to the bottom and accumulate. With them, water is trapped in the interstices between the particles. Initially, while the sediment is at or near the sediment-water interface soon after deposition, there is relatively free exchange between the water occupying the interstitial spaces and that in the estuary above. Addition of only a few millimeters of new sediment to the top of the pile, however, strongly inhibits the free exchange of interstitial water within the sediment layer that is now several millimeters beneath the sediment-water interface. With increasing depth of burial, exchange with the water column becomes increasingly difficult. Bacterially mediated decomposition of organic matter in the sediments proceeds utilizing oxygen as an electron acceptor at the sediment-water interface (1). Within a very short distance below the interface, oxygen is depleted and the bacteria switch to sulfate as an electron acceptor (2) (Richards, 1965).

$$(CH20)_x(NH_3)_y(H_3PO4)_z+x0_2=x(HC0_3^-)+y(NH_4+)+z(HP0_4^=)+(x+2z-y)H+ \quad (1)$$

$$2(CH_20)_x(NH3)_y(H_3P04)_z+x(S0_4^=) = 2x(HC0_3^-)+x(HS)+2y(NH_4^+)+2z(HP0_4^=)$$
$$+(x+4z-2y)H^+ \quad (2)$$

These reactions produce anoxic reducing conditions in the sediment,and the reaction products are released into the interstitial waters where they may reach high concentrations due to the restricted exchange with the open water of the estuary.

Sediment subjected to weathering, erosional and transport processes, is generally exposed to oxic conditions. Thus, the assemblage of minerals in the sediment entering the estuary is adjusted to an oxidizing environment. When this new sediment settles to the bottom and is buried, it is exposed to reducing conditions generated by the reactions described above. Many of the mineral phases are not stable under the new conditions (i.e. iron and managanese oxides) and dissolve. This liberates reduced iron and manganese to the interstitial waters and, in addition, it frees other constituents such as trace metals that commonly are associated with oxides of iron and manganese. The net result of these reactions is to increase the concentrations of dissolved constituents in the interstitial waters. Concentrations increase until saturation is reached with respect to new mineral phases which are stable in the reducing environment. These phases then control the concentrations of dissolved constituents in the interstitial water (Troup, 1974; Bray, 1973; Elderfield, 1979). Generally, the solubilities of the new minerals formed under reducing conditions are considerably larger than the solubilities of the phases stable under oxidizing conditions, so that the concentrations of dissolved species in the interstitial waters are higher then corresponding concentrations in the estuarine waters above. Not uncommonly, concentration differences of 100 fold to 1000 fold are observed (Bricker and Troup, 1975; Bricker et al, 1977). The concentrations of dissolved species in the interstitial waters are controlled in many cases by equilibrium reactions between the mineral phases and the aqueous medium in which they are bathed. (In certain cases where the rate of transport of dissolved species through the sediment is larger than the rate of reaction between interstitial waters and sediment minerals, equilibrium will not be achieved). It is thus often possible from a knowledge of the mineralogy of the bottom sediments to predict the composition of the interstitial waters, or conversely, from a knowledge of interstitial water chemistry to predict which reactive phases are present in the sediment. Interstitial waters are the medium through which reactions occur in the sediment, they are the waters in which benthic infauna are constantly bathed, and they are the vehicle for the transport of dissolved substances through the sediment and across the sediment-water interface. The capability of predicting interstitial water composition is thus a powerful tool in understanding the chemistry of the sediment system and assessing its impact on the estuarine environment. One practical example relates to the practice of dredging. In the Chesapeake Bay dredging for channel maintenance is frequently necessary because of the high sedimentation rate in this estuary. The act of dredging releases the interstitial water contained in the disturbed sediment directly into the estuarine waters. Usually the major effect is simply the release of nutrients which have minor impact in the immediate area of dredging. Knowledge of the interstitial water composition, the water content of the sediment, and the volume of sediment disturbed permits calculation of the amount of dissolved substances released during the operation. In locations where highly contaminated bottom sediment is to be dredged, however, the release of interstitial water may have a significant detrimental effect. An assessment of the potential impact can be made from knowledge of the interstitial water chemistry as above. On the basis of this type of analysis, certain specialized methods of dredging may be indicated, or the dredging may be scheduled for a time of the year where biological impact will be minimal. The other problem with dredging is deciding what to do with the dredge spoils. Many "clean" natural estuarine sediments contain several percent of iron sulfides and other reduced iron minerals (vivianite, siderite). Placing these spoils in a location where they are exposed to oxygen or oxygenated water will lead to the oxidation of the reduced iron minerals, the formation of hydrous iron oxides and the release of a large amount of hydrogen ion to the local environment. The increase in acidity, in turn, may lead to mobilization and transport of trace metals. The mineral-water reactions are completely analogous to those that produce acid mine drainage in coal measures. Again, prediction of sediment chemistry from interstitial water compositions will indicate potential problems of this type and will be of aid in decisions regarding placement of dredge spoils.

Predictions based on sediment interstitial water equilibrium provide valuable insights into the chemistry of the estuarine sediment reservoir and have many practical applications such as the dredging example cited above. This type of approach provides a method for assessing the contribution of nutrients and trace metals from bottom sediments due to direct physical disturbance of the sediment, either by natural processes such as storm resuspension or by man's activities. In addition to these episodic events, there is a continuous and regular migration of dissolved substances across the sediment water interface as a result of physio-chemical processes. In the absence of significant disturbance of the sediment, diffusion gradients develop in response to the concentration differences between the dilute estuarine waters and the more concentrated interstitial waters (Fig. 2). These gradients result in a flux of dissolved substances from the sediment into the estuarine waters above. The magnitude of the flux from the sediments can be estimated using the observed concentration gradient and Fick's first law of diffusion.

$$J_o = - \phi D \left.\frac{dc}{dx}\right|_{x=o} \qquad (3)$$

Where:

J_o = diffusional flux at sediment water interface in $\mu M/cm^2/yr$

ϕ = porosity at interface

D = molecular diffusion coefficient (cm^2/yr)

$\frac{dc}{dx}$ = Interstitial water concentration gradient for the dissolved species of interest as a function of depth in the sediment

Estimating fluxes for Chesapeake Bay sediments in this manner typically yields values within the ranges shown in Table 1. The concentrations of dissolved constituents in the interstitial waters are higher and the gradients are steeper in the summer months than during the winter, implying larger fluxes from the sediments during the warmer part of the year. Biologic activity is also greater during the warm season and probably accounts for the observed increases in dissolved constituents.

Flux of the material from the sediment can be directly measured under certain conditions. The most common method currently being used is to isolate a known area of the bottom under a bell jar or dome type of enclosure. The water contained within the dome is sampled repetitively over a period of time and changes in the concentrations of dissolved constituents are related to rate of flux from the bottom sediments. A major problem in this type of measurement is placement of the dome on the bottom without disturbing the sediment. If the sediment is disturbed, the normal flux rate will be upset and measurements of concentration changes within the dome will not reflect the real natural system. In addition, if the dome is left in place too long, changes in the composition of the water within the dome may become large enough to influence the benthic flux rate. Another problem arises in sampling the water within the enclosure. The amount of sample withdrawn must be small relative to the total volume, otherwise enhanced flow from the bottom into the dome may be induced. Many other technical problems are associated with this type of measurement; however, it appears that with careful manipulation, measurements that are reasonably representative of the natural system can be made. In the Chesapeake Bay, the few actual measurements of benthic fluxes made using the dome technique fall into two categories. Either they correspond well with the fluxes calculated from the interstitial water concentration gradients or they are larger than the calculated fluxes. A number of factors could enhance the rate of transfer across the sediment-water interface. One, in particular, seems to play a major role in augmenting benthic fluxes in Chesapeake Bay. Figure 3 shows an x-radiograph of a two centimeter thick slice of sediment from a box core collected in the upper Chesapeake Bay. It can be seen that from the surface down to the bottom of the core, (40 cm) there is little disturbance or mixing. Layering of the sediment is sharply defined and features such as cross bedding within the layers are well preserved. Benthic fluxes calculated from concentration gradients generally agree well with measured fluxes in this type of sediment. Figure 4 shows a radiograph of a 2 cm thick slice of box core collected in the mid-Bay region. Some remnants of layering can be observed in the upper few centimeters of this core; however, the bulk of the core is very strongly bioturbated by benthic infauna. Most of the features visible on this radiograph are worm burrows, feeding trails, clam burrows and other biogenic structures produced by the activities of benthic infauna. Benthic fluxes calculated from concentration gradients are generally smaller than measured fluxes in this type of sediment. It is not uncommon to find measured fluxes exceeding calculated fluxes by a factor of 2-6 times. The benthic infaunal communities have a profound effect on transport of material in the sediment and across the sediment-water interface. Constant reworking of the sediment by these organisms exhumes material from deep wihtin the bottom and deposits it at the surface where it is once again exposed. The burrowing and feeding activities create open tubes and tunnels in the sediment that serve as conduits for the movement of water into and out of the sediment. The bioturbating activity of the organisms generally tends to increase the porosity of the sediment. The total effect of the benthic community is to enhance the transfer of material across the sediment-water interface (Aller, 1977).

Summary

The massive reservoir of materials contained in the bottom sediments of estuaries have largely been ignored as a potential source of nutrients and trace elements until recent years. On the basis of interstitial water chemistry investigations, it is apparent that there is a very substantial contribution of these substances from the sediment to the water column (Berner, 1979; Bricker and Troup, 1975). Fluxes calculated from concentration gradients indicate that $NH4^+$, HCO_3^-, HPO_4^{-2}, $H_4SiO_4^\circ$, Fe^{+2} and Mn^{+2} are supplied to estuarine waters from the sediment (Table 1). Work is presently being done to quantify these fluxes and to examine other substances, particularly trace metals. For instance, preliminary calculations on NH4+ in the lower Potomac estuary suggests that approximately 60% of the total nitrogen sedimented to the bottom is remobilized into the estuarine waters (Callender, personal communication).

This work at present is in a relatively elementary stage; however, a number of broad generalizations are beginning to emerge. The sediment environment is very different from the freely circulating estuarine waters above. Chemical reactions in the sediment, driven by bacterially mediated decomposition of organic matter, create anoxic conditions within a short distance beneath the sediment-water interface. In this partially restricted environment many nutrient species and trace metals are mobilized and their concentrations are controlled by inorganic reactions (i.e. dissolution and precipitation). These reactions can be described by equilibrium thermodynamic considerations. It is thus possible to predict the concentrations of various dissolved species in the interstital waters, or the presence of reactive mineral phases in the sediment, from a knowledge of the master variables of the system (i.e. Eh, pH, bulk composition). Concentration differences between the interstitial waters and the estuarine waters are generated by the reactions that occur in the sediments. In response to these concentration differences, diffusional transport of dissolved substances takes place within the sediment and across the sediment-water interface. Benthic fluxes calculated from interstitial water concentration gradients using Fick's law compare well with measured fluxes in areas where the sediment is undisturbed by physical or biological processes (i.e. storm resuspension, bioturbation). In sediments that have a flourishing benthic infaunal community, calculated flux rates are generally lower than measured flux rates because of the enhancement of material transport due to effects of the physical activities of the benthic organisms on the sediment. Benthic fluxes calculated from interstitial water concentration gradients thus give minimum values for contributions of material to the water column. Based even on these minimum values, however, it is apparent that the bottom sediments are a non-trivial source of material to the estuary.

TABLE 1

Fluxes of Selected Dissolved Constituents from Chesapeake Bay Sediments. (UM/CM^2/yr)

	HCO_3^-	NH_4^+	HPO_4^{-2}	$H_4SiO_4^\circ$	Mn^{+2}	Fe^{+2}
Northern Bay	770-1310	91-314	0.73-2.19	16.79-28.47	7.30-34.7	9.49-45.99
Zone of Turbidity Maximum	124-325	19-95	1.46-4.02	4.75-25.92	0.37-1.46	1.83-19.71
Southern Bay	390-1160	68-277	5.48-24.82	32.85-40.15	4.01-6.21	0.37-6.57

Figure 1

Simplistic schematic diagram showing the various components of the estuarine system.

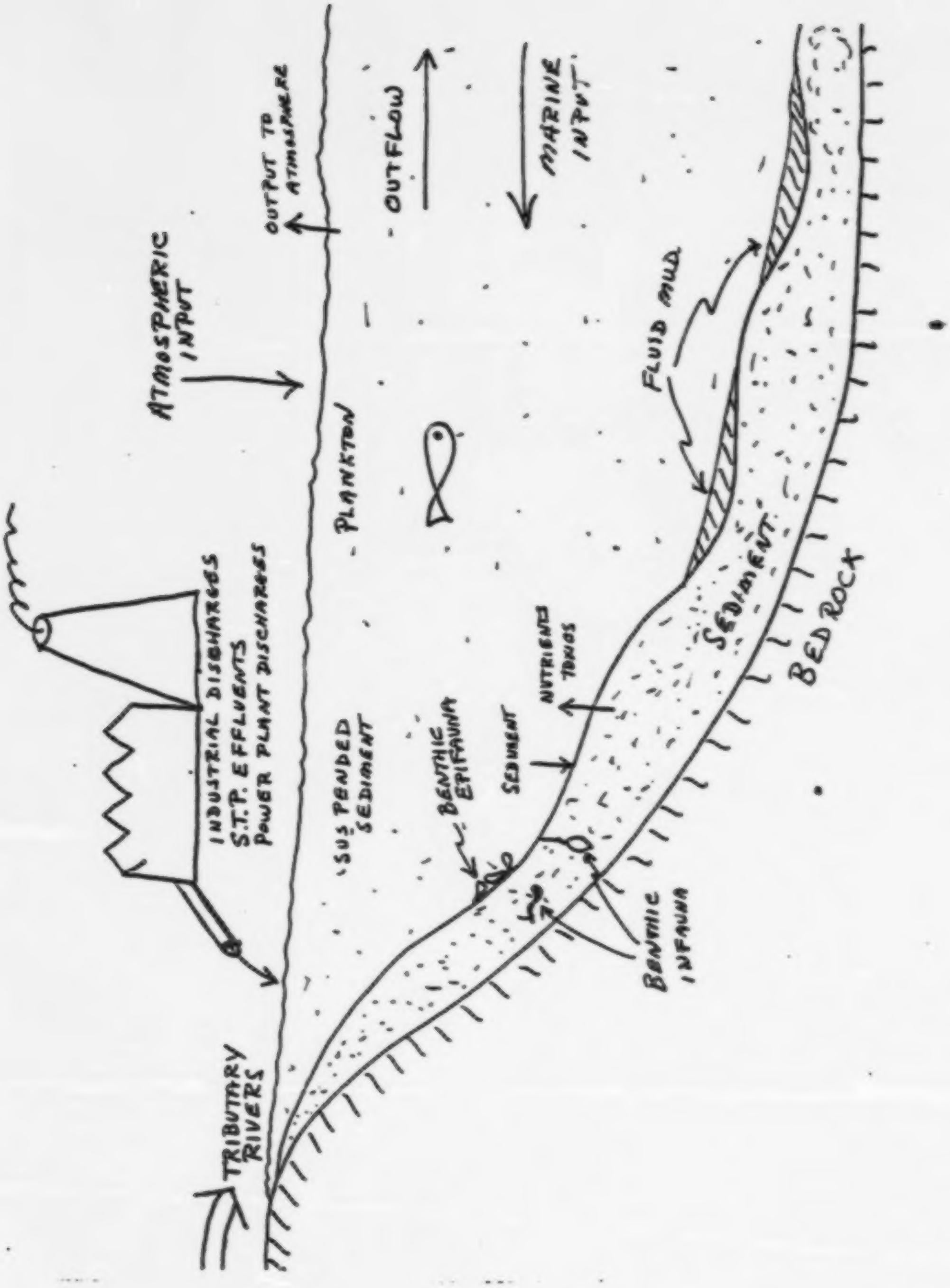
TRIBUTARY RIVERS
INDUSTRIAL DISCHARGES
S.T.P. EFFLUENTS
POWER PLANT DISCHARGES
ATMOSPHERIC INPUT
OUTPUT TO ATMOSPHERE
OUTFLOW
MARINE INPUT
PLANKTON
SUSPENDED SEDIMENT
BENTHIC EPIFAUNA
SEDIMENT
NUTRIENTS
TOXICS
FLUID MUD
BENTHIC INFAUNA
SEDIMENT
BEDROCK

Figure 1

Figure 2

Concentrations of reactive phosphate (solid curves) and ferrous iron (dotted curves) in the interstitial waters of the Chesapeake Bay sediments from the Susquehanna Flats (922) to Wolf Trap (724). Station 856-2 is the shallow water equivilant of 856-8 and Station E.B. is in the center of Eastern Bay.

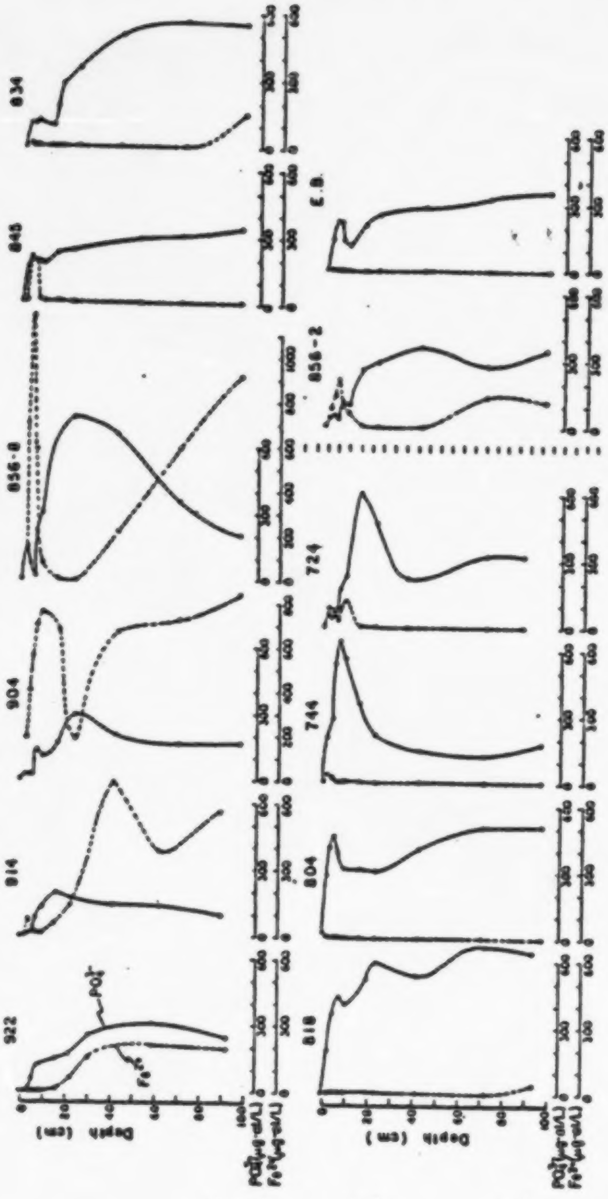
922
914
904
856-0
845
834
818
804
744
724
856-2
E.B.
PO$_4^{3-}$
Fe^{2+}
Depth (cm)
PO$_4^{3-}$ (µg-at/L)
Fe^{2+} (µg-at/L)
0
20
40
60
80
100

Figure 2

Figure 3

Radiograph of box core from Susquehanna Flats in upper Chesapeake Bay. Numbers on right side indicate centimeters beneath sediment-water interface. Horizontal line at 25 cm is match line joining radiographs of upper and lower portions of the core. Irregular lines (a) are shrinkage cracks. Core displays sharply defined sediment layering (b) and well preserved cross bedding within layers (c). Courtesy of Mr. Eli Reinharz, Maryland Geological Survey.

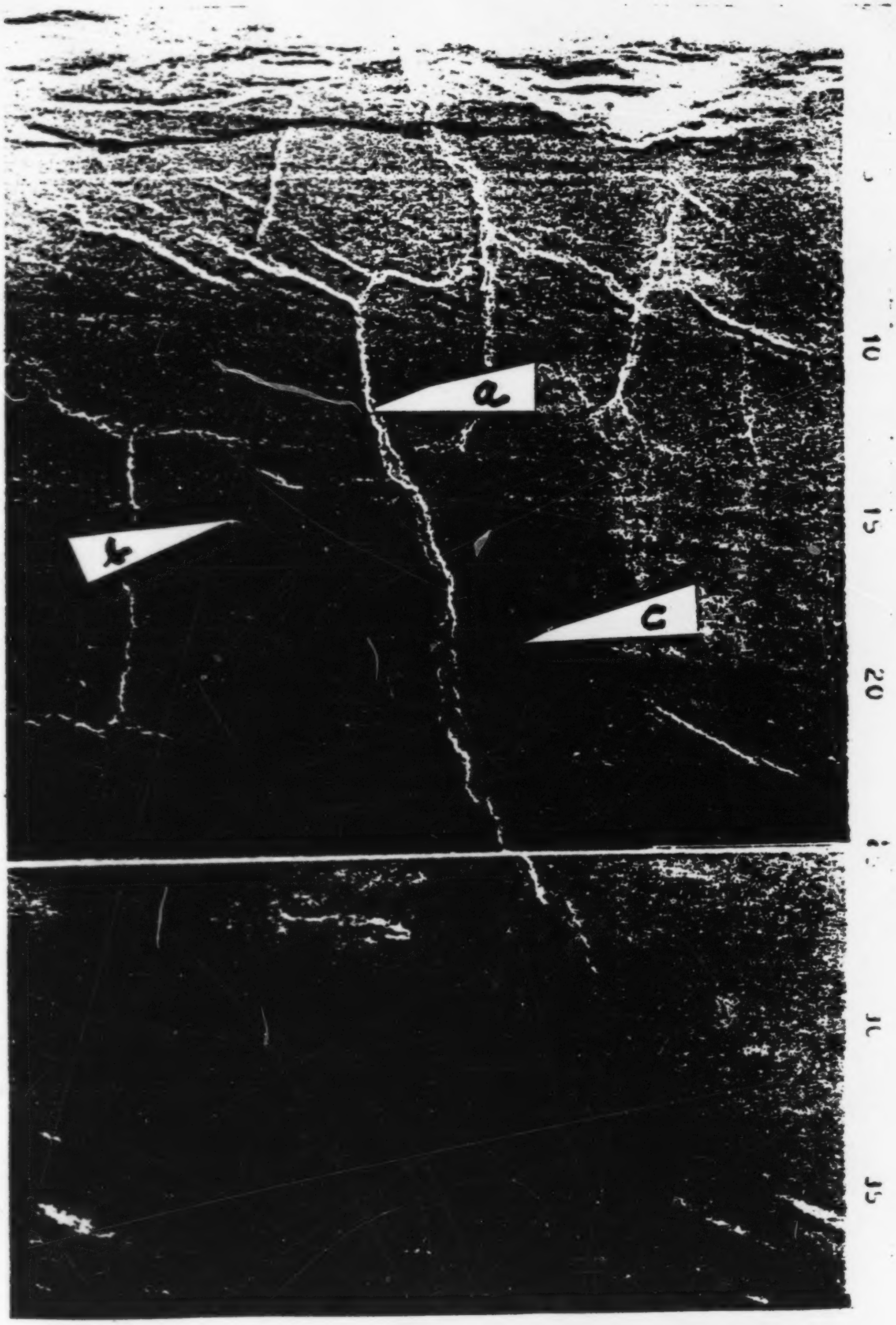

Figure 3

Figure 4

Radiograph of box core from mid-Chesapeake Bay. Numbers on right hand side represent centimeters beneath the sediment-water interface. Some remnants of sediment layering (a) can be seen in the upper few centimeters of the core. All other features are biogenic structures resulting from the activities of benthic infauna. Vertical burrows (b) are clam burrows. Irregular meandering burrows (c) are worm burrows. This core shows very extensive bioturbation by benthic organisms. Courtesy of Mr. Eli Reinharz, Maryland Geological Survey.

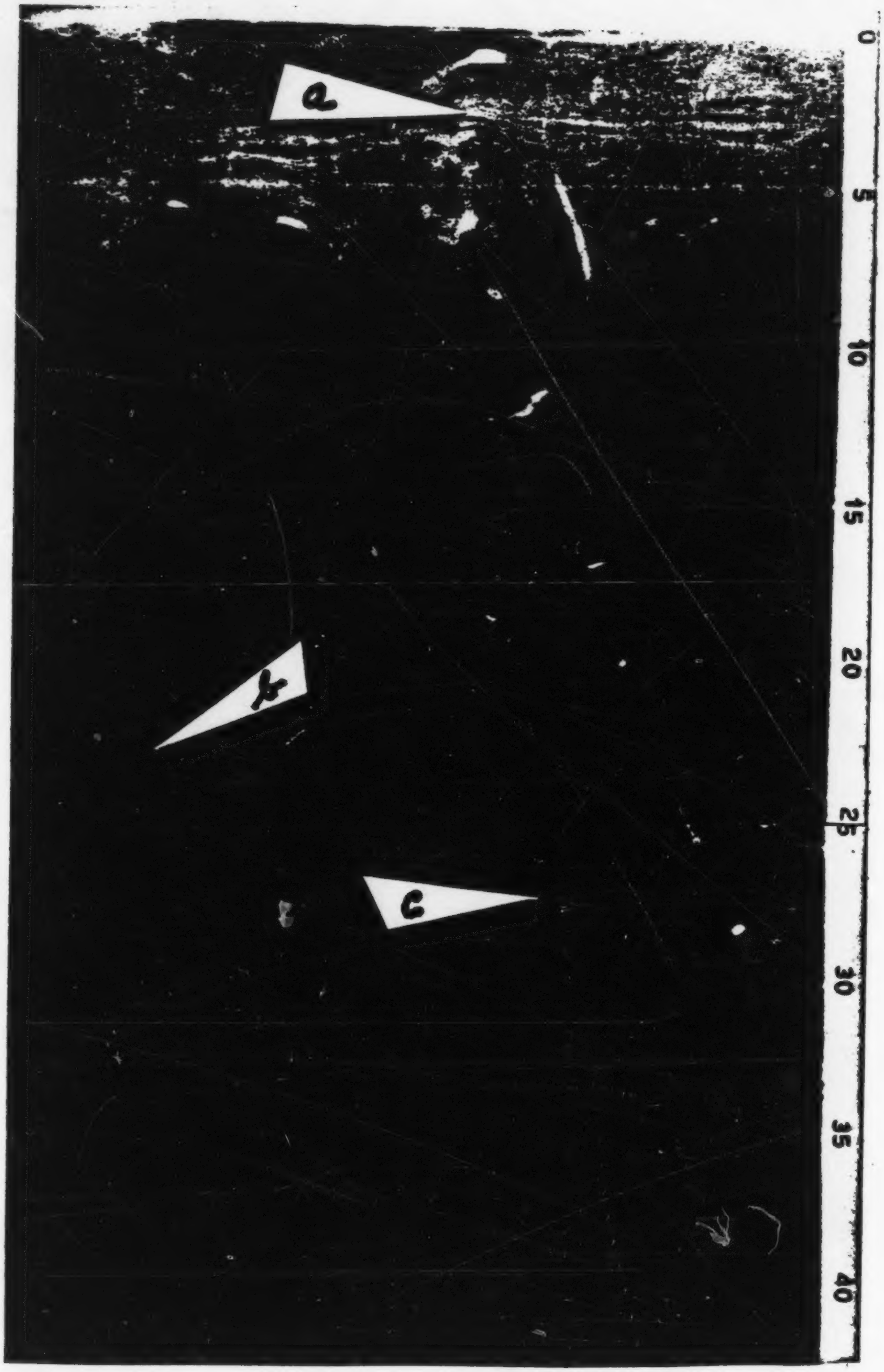

Figure 4

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FUTURE OF CRITERIA AND STANDARDS

SECTION VII

WHERE DO WE GO FROM HERE?
AN INDUSTRY VIEWPOINT

MICHAEL B. LAGRAFF
CORPORATE ENVIRONMENTAL CONSULTANT
STANDARD OIL COMPANY (OHIO)
CLEVELAND, OHIO

The use of water quality standards as a tool to clean up and maintain the quality of our Nation's water as envisioned in The Federal Water Quality Act of 1965 has never been given the opportunity to function. This legislative approach will work if the Environmental Protection Agency will accept the roles defined for it and permit the states to exercise their primary roles. The states must accept this role and provide the necessary resources to accomplish both development and implementation of water quality standards. Industry and other involved or interested parties must participate and cooperate in the standards setting process. EPA is the critical element and how it acts will determine where we go from here--either a cooperative approach is achieved or the Congress and the Courts will tell us where to go.

A. L. Buikema, Jr.
University Center for Environmental Studies
Virginia Polytechnic Institute and State University
Blacksburg, Virginia 24061

My concern is with the development of water quality criteria and our inability to predict ecosystem effects. I am pessimistic about the development of water quality criteria, much less their use in setting standards. My pessimism stems from three areas of concern: a) an inadequate data base; b) manipulation of this inadequate data base; and c) predicting ecosystem effects from single species tests.

There is a paucity of published toxicity data. Of the data available, many are useless because of inappropriate experimental design. For example, most of our animals are tested in "unnatural" conditions, e.g., testing amphipods in substrate free beakers under a high light intensity. All too frequently, the biology of the test species has been ignored or is not known. Toxicity data are obtained by continually exposing an organism to a fixed toxicant concentration for 96 hr while in nature animals may only be exposed to high toxicant concentrations for brief periods of time, e.g., episodic spills.

The last set of water quality criteria published in the Federal Register were manipulated in so many ways that the resultant criteria were scientifically indefensible. For example, the proposed criteria had prediction errors as high as 165%. Most geological benchmark stations did not meet the criteria for metals, e.g., copper. The new method under consideration (Draft, January 1980) results in some improvement. Maximum concentration values may be increased substantially, e.g., copper, but the 24 hr average values may still be over protective. The result may be the unnecessary expenditure of funds to install treatment facilities that will not result in any additional benefit to the receiving water. Also there are aspects of the new method which may not be scientifically defensible, e.g., combining freshwater and saltwater data.

Finally, and most importantly, it is now widely accepted that single species toxicity test data may only be useful to predict gross, e.g., lethal, ecosystem effects. Data for single species, even if they represent an array of taxa and/or functional groups, will not be useful in establishing water quality criteria. Community function and structure is the result of biotic and abiotic interactions and the nature of the interactions is site specific. Laboratory studies do not consider these interactions. The establishment of defensible criteria to protect aquatic life will require studies on complex communities, either in the field or in microcosms where these interactions can occur. Because of wide differences in the ability of receiving waters to assimilate wastes these studies will probably have to be conducted on site.

WHERE DO WE GO FROM HERE?

Paul W. Eastman

Interstate Commission on the Potomac River Basin
1055 First Street
Rockville, Maryland 20850

ABSTRACT

The paper very briefly discusses several reviews and evaluations of water quality standards and criteria recently concluded or currently underway by groups within the American Fisheries Society, the Association of State and Interstate Water Pollution Control Administrators, the Water Pollution Control Federation and the American Academy of Environmental Engineers. Drawing from this information, the author suggests work for the future in the development and use of standards and criteria in order to increase their value in water quality management programs.

WHERE DO WE GO FROM HERE?

This symposium and the question posed to our panel could not be more timely. Several national organizations with which I am involved or acquainted are engaged in activities with the objective of improvement in the development, use and value of water quality criteria and standards. All of these efforts have culminated, or soon will culminate with constructive recommendations about where we should go from here.

The first was a review of the USEPA Quality Criteria for Water, the famous or infamous Red Book during the period 1977-78 by the American Fisheries Society (AFS, 1979). It contains many useful comments and recommendations about specific criteria, and also of particular interest to me, some general comments. Among the latter were concerns about deficiencies in the Red Book data base, the lack of guidance by the USEPA for the development of standards from criteria, and the definition and use of the word, "criterion". All of these comments are suggestive of work to be done in the future. Relative to the last of the previously mentioned general concerns, the AFS review concluded with the opinion that although a collection of national criteria is desirable, "...a single set of national water quality 'standards' would be inappropriate..." Not only AFS, but almost everybody, it seem, feels it essential to eliminate the confusion about definitions of criteria and standards and their respective use, or should I say misuse?

This is not a new problem, however. It has persisted for years. While preparing these remarks, I discovered in my files some testimony which I wrote for a friend in 1965 for hearings on the first federal legislation to include provisions for water quality standards and criteria (Water Quality Act of 1965, P.L. 89-234). I found statements such as, "The term 'water quality standards' is used and misused increasingly as public awareness of pollution grows", and "...information [on which criteria and standards are based] must be applied with caution in the light of the special characteristics (e.g., time of flow, mixing, possible synergistic or antagonistic effects, methods of analysis, etc.) of the body of water involved." Surely we can do better with the cure of problems in the development and use of standards and criteria during the next fifteen years than we have done during the past fifteen years!

Because of the states' deep concern about the subject, the Association of State and Interstate Water Pollution Control Administrators (ASIWPCA) appointed a Task Force on Water Quality Standards in 1979. The task force charge was to determine what the problem areas might be in the legislative, administrative, and regulatory elements of the water quality standards program and to recommend improvements.

The task force first conducted a survey of the states by mail to determine the status of the water quality standards review process in each state. Remarkably, all fifty states responded, which is indicative of their deep concern about the subject. After discussion at a general ASIWPCA meeting and subsequent task force meetings, the task force submitted a draft report to an ASIWPCA meeting in January 1980 (ASIWPCA, 1980) and a final draft is due this month.

One interesting general conclusion is that the states feel there is no need to recommend at this time legislative changes in the goals or stream standards references in Section 303(c) of the Clean Water Act. To the contrary, the emphasis suggested by the task force is on constructive improvements in the oversight-regulatory-administrative aspects. Recognizing that no quick-fix is possible, it recommended that a USEPA-State task force be established to assess opportunities and make recommendations for improving the standards process. A step in that direction for the future took place a week ago (June 18-19, 1980) when the ASIWPCA task force met with a group of USEPA headquarters personnel in Washington, D.C.

The ASIWPCA task force also suggested short-term rotating assignments of state personnel to the USEPA to aid in this process. To USEPA's credit, something of this nature started last fall. Dr. Daniel Sheer of my agency was assigned to work with USEPA last winter on a number of subjects, including particularly the provision of advice on how standards and criteria should be incorporated into the overall USEPA water quality management program. He found a general lack of understanding of how the various pieces of the overall USEPA program should fit together and stressed the importance of a functional structure for the technical tasks of water quality management, including the use of standards and criteria, in organizing a workable program(Sheer, 1980). The part of the functional structure which involves the use of standards and criteria is shown in Figure 1.

During this same period, the Water Pollution Control Federation's Government Affairs Committee established a number of task groups to develop position papers on subjects of vital significance to the national water pollution control program. The Task Group on Water Quality Standards and Criteria, which I chair, is directing its efforts first to a more rational use of standards and criteria in the overall water quality management program. Many feel, as was confirmed by Dr. Sheer, that this is a critical need for the immediate future. Using the Sheer proposal as a point of departure, we expect to have a draft position paper for consideration at the Federation's Annual Conference in October.

Another national group which is actively searching for answers is the American Academy of Environmental Engineers (AAEE). The Academy's Ad Hoc Committee on Concepts for Developing Environmental Standards submitted a report last year which reviewed a mass of information about the underlying philosophies, principles, and approaches used in, or advocated for development of environmental quality standards, particularly as related to three major federal environmental quality control acts (Clean Water, Safe Drinking Water, and Clean Air Acts, as amended)(AAEE, 1979). The report recognized the uncertainties surrounding benefits, risk estimates and costs around which standards are based, and also the unquantiable factors which influence the outcome. It recommended that "The AAEE should promote as its contribution towards developing more rational allocation of our national resources an active program towards identifying value judgements, expressed as benefit-cost ratios, where not explicitly stated in the documentation for proposed or existing environmental quality standards," and determine if there is "...a degree of consistency or trend in the benefit-cost relationships." It also proposed that the AAEE should carry out such studies with teams of its members on a limited number of standards after first making certain to avoid duplication of related efforts by other organizations and agencies. As a member of that committee I support its recommendations. Taking into consideration the uncertainties and unquantifiable factors involved, the benefit-cost relationships of existing and future standards for water and other media should be determined to the best of our collective ability and some degree of consistency within a single medium, e.g., water, and among the different media should be sought and maintained. Although some of this kind of work has been done, much remains to be done in the future.

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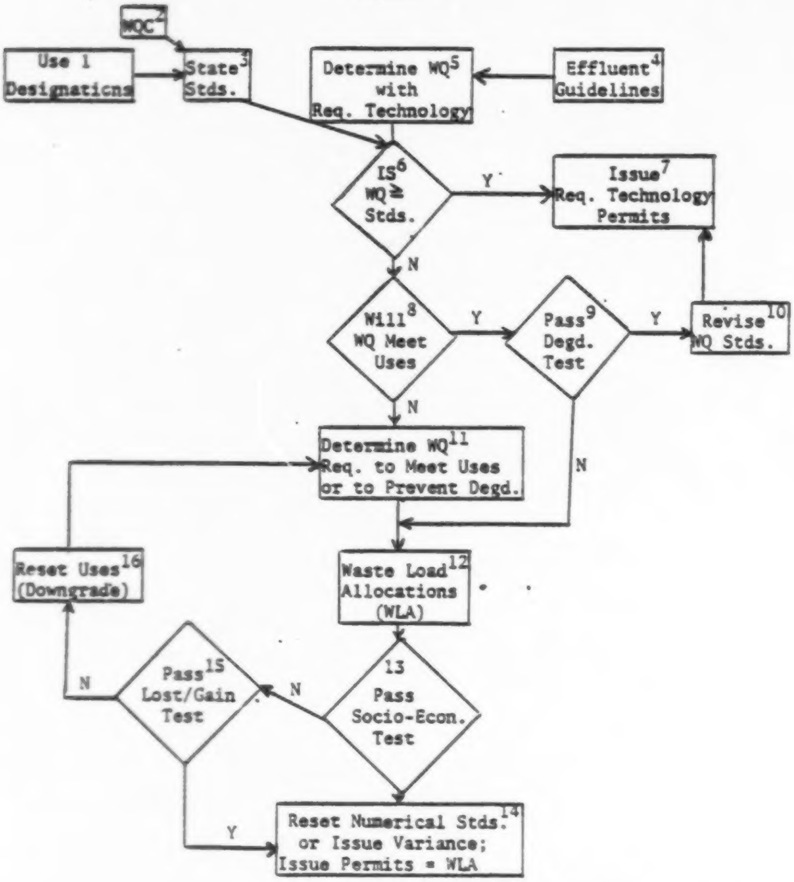
PARTIAL STRUCTURE DIAGRAM FOR EPA WQMP

Figure 1


WQC 2
Use 1 Designations
State 3 Stds.
Determine WQ 5 with Req. Technology
Effluent 4 Guidelines
IS 6 WQ ≥ Stds.
Y
Issue 7 Req. Technology Permits
N
Will 8 WQ Meet Uses
Y
Pass 9 Degd. Test
Y
Revise 10 WQ Stds.
N
Determine WQ 11 Req. to Meet Uses or to Prevent Degd.
N
Reset Uses 16 (Downgrade)
Waste Load 12 Allocations (WLA)
N
Pass 15 Lost/Gain Test
N
13 Pass Socio-Econ. Test
Y
Reset Numerical Stds. 14 or Issue Variance; Issue Permits = WLA

From A Proposal for Structuring the EPA Water Quality Management Program, by Sheer, Daniel P., ICPRB April 1980

AFTER SESSION VII
Floor Opened for Questions

Paul Eastman - Interstate Commission on the Potomac:

I haven't been known as being a great defender of EPA, but I feel compelled to speak up here. I don't think we can blame all the short comings of what we have been doing in the United States on EPA or its predecessors. I think that there is no such thing as an ultimate scape goat in this sort of thing and I think we're wasting our time when we try to find an ultimate scape goat. I'm impressed by the number of EPA speakers here and I think there appears to be at least a rational and reasonable approach being taken with regard to the development of criteria documents. Now, I haven't the slightest idea as to whether these criteria documents, which I understand are going to begin to come out in September, will incorporate what these people have been speaking about or what they've been doing. However, when they're published they don't make the decision on what is published, nor do they make the decision on EPA's policy with regard to its utilization or with what goes into these documents. Those who do make these decisions are the ones which I had pointed out before can be educated. I don't think you can depend entirely upon EPA staff people to educate people. They're not making these kinds of decisions. Nor can you depend upon the EPA staff to educate the people on Capitol Hill who write the legislation which provides the legal framework. I think it's the responsibility of all of the scientists and marginal scientists (I think of the engineers as marginal scientists). I'm not saying that in any derogatory sense. I guess looking for scape goats is not helpful and I think even if we were looking for them, we would find that we're among them.

Mike Lagraff - Standard Oil Company (Ohio):

I just had to cross comment. There is so much criticism of EPA in this regard that I think its the body of EPA and not the individuals within EPA. There are very serious attempts to do what is right even though the individuals who aren't doing what is right think they're doing what is right. That's not a criticism. It's a tough job, having sat in Washington myself, to be given an order and told to carry it out. But I will respond to one point in this regard and that is, I went to a Scientific Advisory Board meeting where the criteria documents were developed. The EPA stood up at that meeting and said we couldn't do our job because the court ordered us to do it in this amount of time and we don't have the data. And I've got to agree. Of course they don't have the data, therefore, they've got to respond to a system that deals without any data. But they did agree to that time frame. They did not go into court and tell the judge that this is an impossible task. Someone accepted this task for the engineers and forced them to do it. So I guess it's the EPA who has to get a perspective of what's going on in the world. I think it's more of a policy issue. It's an EPA issue rather than an individual's issue.

Steve Legore - Hazleton Environmental Sciences Corporation:

I totally agree that the issues are primarily managerial and policy oriented. I have a great deal of respect for many of the scientists in EPA and I hoped that there would be at least one from EPA who does deal with the policy here.

Charles Bostater - Maryland Water Resources Administration:

I'd like to address this question to Mike Lagraff. You just mentioned in your last statements that EPA has not had an adequate data base for development of the criteria documents. Of course, State implementation of standards are based on these criteria documents. Earlier in your talk you mentioned that you feel standards and criteria are the way of the future. Therefore, at one point you are saying that is how everything will work and the next minute you recognize the fact that criteria documents for the standards program are inadequate. Can you elaborate further?

Mike Lagraff:

We were trying to limit our talks to less than 20 minutes and what you asked would have taken considerably longer. There is a gap between where we are today and where we need to be in the future. Your time frame is one of the questions that I have. What is your time frame? Is it tomorrow or is it ten years from now? We need to get that data. That is one of the points Art Buikema addressed. The points addressed are that we need to develop the criteria, scientifically sound criteria. I don't look at it as a number. I look at it as a body of information. Then, somebody in the State or some other organization takes that body of information and translates it into a useful number. If we don't have the data today for a particular contaminant, as far as the criteria documents go, I think EPA should not issue a criteria document for that substance. What they should do is create the mechanism for generating that data and feed that data into the system.

I'm not a biologist, but one of the critiques developed by one of the consultants who works for API on these documents, was that there is no standard methodology for developing this information. A lot of the information we're using is out of college thesis. The guy developed it for one purpose, and now we're extracting it and using it for another purpose. We don't have a national mechanism for criteria and documentation. We need to to set that up. We need to work towards that. Where we feel we have to set the standard and EPA hasn't produced a criteria, then we've got to go to some of these site specific exercises and try to set a standard. Standards can be trial and error and that's something that keeps getting forgotten here. Once you set a standard, it doesn't have to stay there the rest of your life. You set it, you then go back and talk about the dischargers and those types of things and figure out what the dischargers can do. If the standard works, fine, we've cleaned up the river, we've got the use. If it doesn't work, then let's take another crack at it, let's check our use, let's check our criteria, let's check our dischargers and go back to the process. I don't think there's a conflict between the two. I think it's a time function and a dollar function. I think standards can work in time.

Charles Bostater:

Do you feel that it's important then to leave water quality standards alone and do away with effluent limitations?

Mike Lagraff:

No, water quality standards are the base. From that base, we've got to have planning. We've got to have 208 planning which is a mechanism by which you come back to those standards. You've got to take the standards and go to waste load allocations and the effluent limitations. The standards are the measure that we have. That is, the use we want and the measure we have for that use. It's an integrated tool we've never received and I think that's one of the problems within EPA. 208 planning is on the fourth floor, Water Quality is in another tower, and I don't know where the effluent limitations people are located. We never brought the picture together. I think it should be brought together with the Water Quality Standards either at the top of the pyramid or at the bottom.

Jerald Schnoor - University of Iowa:

A question I'd like to leave open is: I see in the criteria development, the little bit that I know about it, that there are some specific problems with specific numbers. Some of the metals, for example copper, were brought up several times today as not having an adequate data base. There may be some inadequacy in the methodology of the proposed documents, but I think there is at least a movement afoot to correct what is obviously a problem in the methodologies, primarily regarding the metals. I don't see the problem as much in regard to criteria development, which I see as a necessary guide for States in setting water quality standards, as a danger if these if these criteria become water quality standards. This has happened in the past, sometimes due to lazy States and things which Mr. Lagraff pointed out. To the extent that we can use these documents to set site specific, or at least specific water quality standards, I don't see a problem.

Mike Lagraff:

If EPA would do away with the Red Book policy, I don't care what they publish as criteria, as long as the idea is established that standards are set based on what is needed to protect a given use by a responsible State agency. I think Art (Buikema) hit it--it's the fact that is is the only thing we have to use. If there is something wrong, is it bad putting it out like that, without telling us there's something wrong with it? This is what happened within EPA. Maybe there's someone here who was around when the Green Book was written. There was a great debate at the time as to what increment of temperature should exist. I met with a couple of people who worked on the Green Book who explained that they couldn't resolve the issues is it to be a Δ3 or Δ5, so the two of them sat down and said "let's make it Δ5 or Δ3 and get over with this debate". The criteria are only guidelines. New you've got Δ3 and Δ5 numbers written in standards everywhere. That's how it evolved. EPA says something one day and a scientist says here's a good idea, then before you know it, it's a Bible someplace. That's what happens to these criteria documents. They become Bibles.

Tom Felvey - Virginia State Water Control Board:

I would like to comment on a problem a State has. I would agree that I think the criteria are good when they are guidelines. I'm a biologist and refer to them quite frequently. But the problem that comes to us is when EPA forces these on us as a standard. We in the State of Virginia have standards, the same ones most states have, i.e. dissolved oxygen, pH, temperature, etc. We have adopted the EPA criteria, I don't know whether rightly or wrongly. We adopted them as State criteria for our surface waters. We were more or less forced into that situation. At the present time the EPA has put quite a bit of pressure on us telling us that those criteria now are our standards in the State of Virginia, and we say they are on our books as criteria and nothing more. Right now we are undergoing that argument and I think that's where the problem is; when EPA tries to force these on us as actual standards. I think as guidelines they are very good, but not as standards.

Brock Neeley - DOW Chemical Company:

Maybe we could change that name, get rid of that word criteria.

Robert Thomann - Manhattan College:

I guess I've reflected a lot on this whole issue of criteria and standards, public participation, setting of uses and so on, and I've seen the pendulum swing back and forth over the last two decades. But I think in recent years, the new wrinkle, at least that I've lived through, is the introduction of litigation as a way of solving problems. And I wonder what the panel's reaction is to the following kind of comment that I hear not infrequently. What you're proposing is very sound: set water uses, waste load allocations, carry out trade offs between cost and benefits. That's all very nice. Operationally, it's rather hopeless because it really doesn't work that way. Criteria are uncertain, credibility of the analysis are uncertain and the regulatory agencies just don't like to accept that entire framework, especially in today's environment of contemporary chemicals which may have some toxic effects. So given the fact that operationally it doesn't seem to work, when we really get down to a case to case basis, why don't we just step back, let EPA set whatever they think they want to set and then take it to court.

Paul Eastman:

I'd like to comment on that because I was Director of Water Pollution Control for the State of New York and at the time Congress was considering the Federal Water Pollution Control Act Amendments of 1972. We had a group of State administrators who, on an ad hoc basis, met in Washington from time to time. We occasionally had legal help, that I thought was helpful at the time. We had very little influence on what came out in 1972. I remember one of the arguments for technology based effluent requirements was that there could be an expediting of accomplishment because there will be less uncertainty and less resort to the courts. In fact, you might call it the lawyers relief act of 1972 because it greatly increased the employment opportunities for lawyers, which is not all bad, but it led to a need, for example, for a better education of a wider variety of people that are in the milieu of making decisions, the courts. Some judges come out with fairly decent decisions. We have one of the best sludge judges, in I think, all of the United States. A U.S. District Court judge has the unusual name of Smith in the Washington, D. C. Area, a particularly knowledgeable judge, but we're not always that lucky. Getting to the final point in your question, I think, that what should be done is, regardless of what EPA says, and when I say "EPA says" it's meant in meaning whoever is the Assistant Administrator for Water and Hazardous Materials in any point in time. The State's ought to get their act together and do what they think is best for their particular State. Then let EPA take a guess. Now I know some States have done this and the ones that I am familiar with, not intimately, I'm not saying that they took their steps in the most rational, reasonable way, but I think that supposedly the responsibility is primarily with the State. Let them take their action with regard to the best information they have available, establish their standards from whatever criteria documents they seem to have, and anything that EPA provides can be critically evaluated by the States with whatever resources they have in their selection of particular priorities, particular bodies of water, and the rate of implementation of particular standards. Then it may very well wind up in court, but at least in the meantime I think it's going in the right direction. One of the big mistakes I think States have made, and I'm not blaming anyone on this, is blindly accepting the so called Red Book Values. In many cases I believe the States knew what they were adopting and now with chagrin, realizing this. I happened to, for one particular substance, to consider what was in the 1972 National Academy of Science Blue Book which is supposed to be guidance for EPA and their 1974 Red Book and they couldn't have been too far apart from the recommendations in the Blue Book, which I considered a critical evaluation of the State knowledge. It said for this particular substance that "there is no scientific basis for adopting a number", and then a footnote said, "well if a number is essential, we would recommend one thousand." In the Red Book which came out two years later, not only was the number adopted, but it was 1/5 of what was recommended in the Blue Book. What has been adopted by the State is what came out in the Red Book. Now in many cases they're stuck with an unenforceable, extremely difficult standard.

Mike Lagraff:

I think that's what happens when you leave these things in the hands of the judge. You don't necessarily get what you want.

Charles Bostater:

I think that a lot of State people that I talk to have a technical background and the State's don't just go ahead and adopt the Red Book. I believe that there's a lot of careful consideration in any use of that Red Book and Blue Book. I think Tom Felvey from Virginia pointed out that they research the relevant information themselves a lot of times. Sometimes State people are more aware of recent literature and other obscure literature that may not be in those Red Books. I think sometimes we have to give the States more discretion and credibility in the criteria and permit area.

Brock Neely:

I'm sure there is a great variability among the States.

Paul Michael - Monsanto Chemical Corporation:

There have been several comments and points that have been repeated in the last few days. One is that the water quality standards and criteria should be site specific and use specific. Also, several people have cited what role the States should play. I just wondered what happens when the common circumstance occurs when a river or a lake pours into more than one state. To use Mike Lagraff's example, suppose the people in the vicinity of a city with steel mills decide they want jobs and don't mind the discharge of significant amounts of material into a river. The people a few miles downstream, who may live in another state, may decide they want to use that same river for recreational purposes. Does anyone have any ideas on how to control that problem?

Mike Lagraff:

That's where EPA's review process comes in. That's where they started out back in 1965. They were dealing with those boundary waters, and what the State did inside the State was the State's business. I thought it was a good act at the time. But there are going to be problems. You're going to run into that all the time. 208 helped do that a little bit where they set up interstate agreements to deal with these areas. The Potomac River Basin Commission is an example of a commission that can work in this regard to try to resolve the issues between several states. No one is saying this thing is going to be easy, we're going to resolve all these issues, and everybody's going to be happy when it's done. But I think it can be worked out.

Peter Hodson - Canada Center for Inland Waters:

You expressed concern that you observed copper concentrations in various rivers in which you were not seeing any copper related effects, and you were concerned that we had an excess degree of protection for those rivers with the copper criteria required. These criteria in general seem to be based on laboratory bioassays where sensitivities were a function of the number of organisms tested, times tested, and how good the experimenter is in developing this data with his techniques. I was wondering whether you think we will ever see in the environment any levels of response that we use to judge what the criteria should be? In other words, lets take a 10% drop in fish egg production. We are never going to see that in the environment distinct from the natural variability. So you could say yes indeed there is a deleterious effect on the environment.

Arthur Buikema - Virginia Polytechnic Institute:

That's a hard question to answer. Given what I feel is the inadequacy of the things you can see in the laboratory, I don't think we're going to see that in the field. I think one of the problems, if I understand the question, is one of having to interpret the significance of some of these tests as well. One of the problems ASTM is having is one of their committees is using MATC values with daphnia. One of the arguments right now is what does 50% impairment of reproduction or 25% reduction mean. My feeling is we will have to stick with a site specific base, looking at the entire community, and I don't know exactly how to do that yet.

Peter Hodson:

In the Great Lakes we have had tremendous changes in fish population. There are supposedly 18 anthropagenic stresses on the Great Lake fisheries. No one has yet been able to point to a single major stress as the cause of decline in the fisheries. Therefore, how can we identify a minor stress as inappropriate, important or not important?

Paul Eastman:

Well, if I could make a little comment on that. I think in a situation in which there are some 18 stresses being placed on the organisms in an environment, there cannot be any differentiation between the relative effect of those 18 stresses. I think the assumption is that there's no scientific basis setting priorities with regard to attempt to control, then we look at the cost of each of those 18. If you find that you can rachet back, say six of them at relatively low cost (not only monetary cost but general social economic effect), then I think that the regulatory agency can move in that direction until more is learned about the sensitivity to the degree that differentiation can be made between the relative effect of those stresses. It seems to me this is the approach they've taken in the United Kingdom. I think they've done a tremendous job in the United Kingdom, particularly on the Thames in improving the quality of water, but we had a fellow who is now the Chief Executive of Thames Water Authority in the U.S. just two years ago. Of course there's been a lot of national and international publicity about the achievements well justified on the Thames and he made the statement several times that the Potomac is cleaner than the Thames. But you certainly wouldn't get that impression if you were around the U.S. It's still one hellishly polluted river and so the tendency is to try to do everything at once on the Potomac. Well, we wind up in court and not everything gets done at once. The approach on the Thames is, I think, extraordinarily good. They try to isolate those things which happened to have the greatest effect at the least cost, and the result is what might be considered at least a tolerable cost. I don't know whether it's a reasonable cost, but they have been able to achieve a great deal without an extraordinary unfavorable impact on the people and the economy.

Charles Bostater:

I'd like to address this to Paul Eastman. You've been in this business a long time and in the Potomac you have multiple uses, perhaps some economic uses. Do you think there's ever been an adequate economic definition of uses of affected systems? We saw slides showing the need to adequately define the uses. Do you see the need to define the uses economically?

Paul Eastman:

Well, when I was with the U.S. Public Health Service, some 200 or 300 people were trying to do something at the Federal level regarding water pollution control. It was decided that we ought to hire an economist. We looked around and found an economist. I had lunch with him the first day of his arrival in the Washington, D. C. headquarters of the Public Health Service, and his immediate response was, this is an enormously complex and difficult situation in which to evaluate from an economist' point of view. He had enough knowledge of a physical, chemical, and biological scientist, and he knew that there was much more data in these fields than in the economist's profession. I think this is still the case. But that doesn't mean that we shouldn't get the best information we can get our hands on. To the degree that we have the resources for that purpose, in order to provide the basis for the judgement by the people who are elected or appointed to make those decisions, we have to be sure that they are provided with the best information concerning the intra and intermedia consequences of their choice of one of the many options. In short, I think we have to provide that information within our ability. Again, this means to set priorities on the resources. Incidentally, in the Potomac, as far as its economic impact is concerned, the shellfish industry does not provide very much to the economy of Maryland and Virginia which share the shellfish resources of the Potomac estuary, but it has its historic and cultural value. Now that may be difficult, if not impossible, to convert into monetary terms. I think that sort of thing has to be put into an equation and I would be the last person to say that we shouldn't do everything possible to preserve the shellfish industry on the Potomac estuary because of its historical and cultural contribution to society. In that connection the Potomac is improving. There are an increasing number of sitings of monsters in the Potomac estuary, which, at least in my lifetime, have apparently been driven away by pollution or whatever. The most recent one was just reported the day before yesterday in the New York Times. It apparently has a diameter of roughly the size of a quart jar and is some eight to fifteen feet in length. The people who sited it said that they absolutely had not had anything to drink, so their credibility cannot be questioned.

Bruce Bell - The George Washington University:

I just wanted to say thanks to everyone who contributed toward this session in terms of speakers and the audience. I found it very interesting, entertaining and unfortuately I think I found out more we don't know than we do. But that's usually the nature of these things. I'd just like to thank you all for coming and have a good trip home if you're heading out. Thank you.